Urbanisation influences on freshwater fish distribution and remediation of migratory barriers

---

A thesis
submitted in partial fulfilment
of the requirements for the Degree of

Master of Science in Ecology

in the University of Canterbury
New Zealand

By
Katharina A.M. Doehring

University of Canterbury
2009

TABLE OF CONTENTS

ACKNOWLEDGEMENTS

I am very grateful to Angus McIntosh (University of Canterbury) and Roger Young (Cawthron Institute) for their supervision and guidance through this study. They have always supported my sometimes not "viable" ideas about how to conduct science and patiently edited my German-structured sentences. I really appreciate their everlasting enthusiasm and speedy responses to my queries and chapter drafts. Time seems to have flown past and I take it as a sign that I really enjoyed working with and learning from both of them.

Special thanks also go to the people of the Freshwater Ecology Research Group (FERG) who have helped me shape my thesis with their constructive feedback and endless patience with statistics and GIS. I will miss being part of this amazing team and wish everyone the best for their research projects.

I would like to thank Jacques Boubèe and Cindy Baker from the National Institute of Water and Atmospheric Research (NIWA) who helped me in the very beginning of my research with the experimental design and execution of my laboratory experiments.

The Nelson City Council (NCC) and the Cawthron Institute supported me financially and provided me with field equipment, advice and comprehensive data about the area (i.e., GIS data), for which I am very grateful. I particularly thank Paul Sheldon and Morag Fisher (NCC) for arranging all the access to field sites and their loyal support while fixing my "car park dilemma". I would also like to thank Aaron Quarterman (Cawthron Institute) for trusting me with the professional equipment and supplying me with his technological knowledge about pumps and video cameras. Also special thanks to many others in the "Glenduan Aquaculture team" for their support, but particularly for the bench space during times where the oysters started to outgrow the nursery facilities.

Martin Rutledge (DoC) and Neil Deans (Nelson Marlborough Fish & Game) not only introduced me to the Nelson Region's waterways and fish fauna, but also provided transport and

field equipment. Thank you to you both, I really appreciate your support. The Nelson DoC Conservancy supplied office space during my writing up. I would like to thank Kerry Brown especially, for putting up with me in his office as well as for all the interesting discussions about absolutely everything – all related to science of course!

Many people assisted with field work, particularly during the first summer, and I truly thank everyone for their help. I would like to especially acknowledge Pam and Graeme McConnell, who not only helped me in the field, with the construction of my experiments and scientific advice, but also gave me a "home" during my research in Nelson. Pam and Graeme, I surely would not have been able to focus so explicitly on my study if it wasn't for you looking after me.

My family's support, encouragement and love – overseas as well as here in New Zealand – have been never-ending and I would not have been able to finish my thesis without them. I would like to thank Yvonne and Donald for making me feel at home in New Zealand over the last two years. I also would like to thank them for their support with field work and scientific advice. Mama and Papa, I thank you for introducing me to the principles of ecology during my early childhood at our bach (Mühlthal) and your encouragement in letting me go overseas to study. Your strong belief in me gave me the strength to get this far. My greatest thanks, however, go to Blair. He not only looked after me during my under- and postgraduate studies (I know I am high maintenance!), but also put up with my 'stress levels' over the last 18 months. You contributed to this research in so many ways and I thank you for your eternal patience and understanding.

ABSTRACT

Ongoing urbanisation in lower stream reaches can adversely affect the ecology of river ecosystems by altering freshwater fish distributions. Diadromous fish species (those that migrate between the sea and freshwater) are particularly affected as unimpeded access to and from the sea is necessary for their life-cycle completion. To investigate the extent of the effects that urbanisation have on migratory freshwater fish species, I compared fish community composition between urban-impacted and un-impacted waterways in the upper South Island, New Zealand. I also investigated whether culverts were barriers to inanga (*Galaxias maculatus*) juvenile migration, and experimentally tested different aspects of fish ramp designs for the remediation of juvenile inanga migration barriers. Analyses of fish community composition indicated significant differences between urban-impacted and un-impacted stream reaches. Fish species sensitive to poor habitat quality were generally absent from urban waterways. Inland penetration of fish species was reduced in urban-impacted than un-impacted streams, likely indicating culverts were upstream migration barriers. Laboratory experiments showed that climbing ability of inanga depended on fish size. Climbing also depended on the length and slope of the climb, but not the availability of resting pools. Experiments conducted at 13 natural culverts showed fish could not pass (0% passage) undercut (perched) culverts unless aided by an experimental ramp (44 % passage). To offer successful upstream passage for weak swimming species, culverts should ideally be box-shaped with widths and angles similar to the natural stream bed. Fish ramps should be designed to accommodate species with weak swimming abilities, considering trade-offs in ramp geometry. My results provide a framework for identifying and correcting barriers to diadromous fish passage, allowing managers and ecologists to maintain and enhance fish populations in urban environments.

Toi Tu Te Marae O Tangaroa

Toi Tu Te Marae O Tane

Toi Tu Toi Te Iwi

"If the domain of Tangaroa (God of water) and Tane (God of the forest) are healthy, so too the people are healthy."

PREFACE

Cities have been a part of human history for millennia and predictions suggest the majority of humans will live in cities in the future (UN 2007). The rapid extension of urbanisation represents a threat to stream ecosystems as stream habitat and biota in urban settings are often profoundly degraded in comparison to natural or less-impacted rural environments (Paul & Meyer 2001; Miltner *et al.* 2004; Walton *et al.* 2007). Given these impacts and the accelerating pace of urbanisation, there has been increasing interest in defining the thresholds of degradation imposed by urban development and the potential remediation of urban stream ecosystems (e.g., Allan 2004; Grimm *et al.* 2008). In 1992, the United Nations Division for Sustainable Development (DVD), for instance, adopted Agenda 21, which describes the "protection of the quality and supply of freshwater resources" at local, national, regional and international levels (UN 1992, page 196). Eight years later, the European Union launched the European Water Framework Directive (EU-WFD), a Europe-wide guideline on how to protect water resources (European Parliament and Council 2000). Both policies include the sustainable management of fish populations within every single waterway in their partner countries.

The demand for "healthy" waterways has also grown in New Zealand. Due to increasing development pressure within the upper South Island, the Nelson City Council (NCC) together with the Cawthron Institute, the Department of Conservation, and the Nelson/Marlborough branch of Fish & Game approached me in spring 2007 with a request to record freshwater fish occurrence within the NCC area to provide an up-to date summary of fish distribution patterns. As little is known about the ways urbanisation affects the distribution and community patterns of New Zealand's freshwater fish species (Suren & Elliott 2004), I extended the initial NCC request to further investigate the effects of urbanisation, and migration barriers in particular, on freshwater fish species. Migration barriers are likely to occur in any city near the coast and are

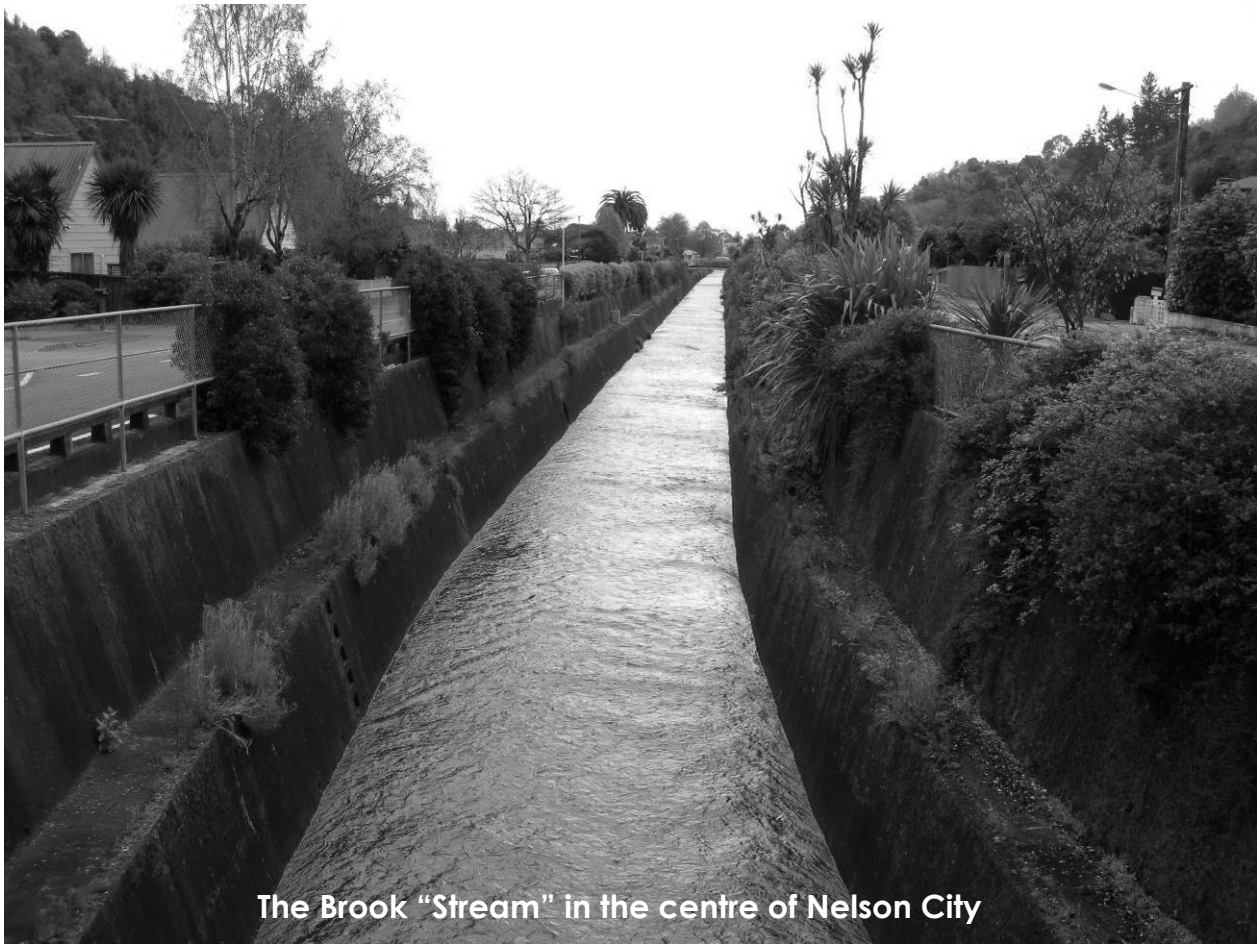
therefore a worldwide issue that needs to be understood to maintain and/or enhance freshwater communities in urban environments.

In this thesis I investigated the effects of urbanisation on freshwater fish distribution and the potential remediation of these effects. Although my experiments were planned for, and conducted with New Zealand fish species, my results can be applied to other cities and to non-New Zealand fish species with similar swimming behaviour and capabilities. My research questions were: what is the current distribution and community composition of freshwater fish species in Nelson City and what habitat variables such as land use or migration barriers influence them (Chapter One)? How can migration barriers caused by culverts be overcome and which key features of fish ramp design need to be incorporated to facilitate upstream passage of juvenile inanga (*Galaxias maculatus*) (Chapter Two)? What factors influence whitebait passage success in relation to various *in situ* culvert designs and how can upstream passage at culverts be improved by ramp installation (Chapter Three)? Finally, based on my findings I summarise how councils and managers can mitigate the effects of culverts as migration barriers, and provide some advice for key aspects of fish ramp design (Chapter Four).

My thesis has been structured as a series of papers that are intended for publication and this has necessitated some repetition.

Chapter One

THE DISTRIBUTION OF FRESHWATER FISH SPECIES IN AN URBAN ENVIRONMENT



The Brook "Stream" in the centre of Nelson City

1.1 INTRODUCTION

Human actions are a threat to aquatic habitats, influencing the ecological integrity of river ecosystems via numerous and complex pathways (Allan 2004). The ongoing spatial expansion of urban land use represents a threat to stream ecosystems worldwide, as stream habitat and biota in urban settings are often profoundly degraded in comparison to natural or less-impacted rural areas (Grimm *et al.* 2000; Miltner *et al.* 2004; Blakely *et al.* 2006; Walton *et al.* 2007). By 2050 70% of the world's population is expected to live in urban areas (UN 2007), so the likely impacts of cities on natural ecosystems are enormous. For example, modern European cities require the ecological services of between 500 to 1000 times the land area they occupy. As many of the required ecological services in cities cannot be met, urban ecosystems, in general, are frequently degraded (Folke *et al.* 1997).

Effects of urbanisation on aquatic ecosystems include increased impervious surface area with increased runoff to receiving streams, higher peak discharges, greater water export and higher loads of fine sediment during the construction phase of, for instance, roads (Hogg & Norris 1991; Nelson & Booth 2002; Bernhardt & Palmer 2007). Over time these effects can adversely affect the ecology, physical characteristics and visual appearance of river corridors, altering the habitat of dependent biological communities (Paul & Meyer 2001; Grimm *et al.* 2008). The dramatic threat urban expansion poses to stream ecosystems and the increasing awareness of the economic and social losses arising from stream degradation due to urban extension has led to an increase in studies focusing on the effects of urbanisation on stream settings (Lake *et al.* 2007). However, what exactly drives freshwater fish species distribution in cities, and to what extent urban fish species community compositions differ from unimpaired habitats, is still poorly understood. To efficiently manage the effects of land-use intensification on freshwater fish communities, improved knowledge of the magnitude of these effects is

needed. In this study I evaluated the ecological effects of urbanisation on freshwater fish species through a case study of the Nelson City Council area in New Zealand.

Most of New Zealand's native freshwater fish are, except for eel species, relatively small (adults generally <300 mm long), cryptic and benthic (McDowall 2000), and despite becoming better understood, relatively little is known about their spawning biology, community structure and specific habitat preferences (Jowett & Richardson 1994; Allibone & Caskey 2000; McDowall 2000). To be able to assess the influence of urbanisation on New Zealand native fish, an understanding of their lifecycles is crucial.

There are 35 native fish species in New Zealand, including 31 (88 %) that are endemic (McDowall 1998, 2001). As 16 (45 %) of these species are diadromous (McDowall 1995), their population and community ecology, as well as their biogeography, are affected by issues such as migration barriers.

Diadromy is a "regular, physiologically mediated [movement] between freshwater and the sea [occurring at] predictable life history phases in species" (McDowall 1998, page 112). There are three types of diadromy (anadromy, catadromy and amphidromy). Once maturity is approached, catadromous fish migrate to sea for spawning, while amphidromous species spawn in freshwater where their larvae, once hatched, move immediately to sea. Anadromous fish (e.g., lamprey, Chinook salmon) enter rivers as mature adults to reach upstream spawning grounds to reproduce (McDowall 1995, 1998). As migration in and out of rivers, as well as the life stages that undertake the migration, differ among species, interpreting differences in the behaviour of native migratory fish is important for understanding their community structure and distribution patterns (McDowall 1998). Although upstream penetration differs for each fish species, finding suitable habitat for growth and reproduction is a common aim for each (McDowall 1998). Hence, studying the movement patterns of successful juvenile recruitment to upper stream reaches is fundamental to understanding their spatial distribution and how adult populations are maintained (Eikaas *et al.* 2005; Eikaas *et al.* 2006).

Fish communities in New Zealand generally differ depending on the distance inland (McDowall 1998; Joy *et al.* 2000; Leathwick *et al.* 2008), so separating "natural" fish distributions from those affected by urbanisation poses a challenge. Waterways close to the sea, as in many cities worldwide, may play an especially important role for migratory native fish species, as they act as corridors to upstream unimpaired habitat, required for reproduction and growth (McDowall 1998). However, it is the lower and intermediate reaches of urban waterways that are especially altered by urban development in New Zealand. Significant changes include, for instance, water abstraction, channelisation and the installation of physical structures, which may result in freshwater fish population fragmentation and significant alterations in species abundance in these areas (Suren & Elliott 2004). Physical constrictions, especially, are a major impediment for natural fish distribution within urban areas by restricting fish movement between habitats (Baker & Boubée 2006). Culverts, for example, can form a boundary, capable of altering local physical stream characteristics, hydraulic conditions and biotic linkages within streams (Peake 2004; MacDonald & Davies 2007). Mitigating impacted urban stream ecosystems is important for the communities they support with freshwater fish playing a particularly important role because they are widely seen as an indicator of life-supporting capacity in rivers (McDowall & Taylor 2000).

In this study I investigated (1) which fish species currently exist in urban stream systems and (2) determined what might influence their community structure by comparing fish species distribution between an urban (Nelson City, New Zealand) and a non-urban area (the upper South Island of New Zealand) in the same region. I also investigated (3) the impacts of culverts as physical migration barriers on urban freshwater fish species distribution patterns.

1.2 METHODS

1.2.1 Study sites

1.2.1.1. The Nelson City Council Area

I chose the Nelson City Council (NCC) area, located on the northern coast of New Zealand's South Island, as a case study to investigate the impacts of urbanisation on New Zealand's freshwater fish distribution. The area encompasses 443 km² stretching from the Waimea Inlet to Cape Soucis, inland to the ridge of the Bryant Range in Mount Richmond Forest Park and along the eastern margin of Tasman Bay (Fig, 1; Crowe 2002). Being one of the largest and fastest growing cities in the South Island (Paul Sheldon, Nelson City Council, personal communication), Nelson City (population size = 42,891; Nelson City Council) likely poses substantial pressure on the city's waterways. Three catchments and eight subcatchments were selected to encompass the variety of land use and topographic conditions within the area including small urban streams (Jenkins, Orphanage, Saxton Streams) and larger catchments, such as the Wakapuaka and the Maitai River Catchments (Fig. 1).

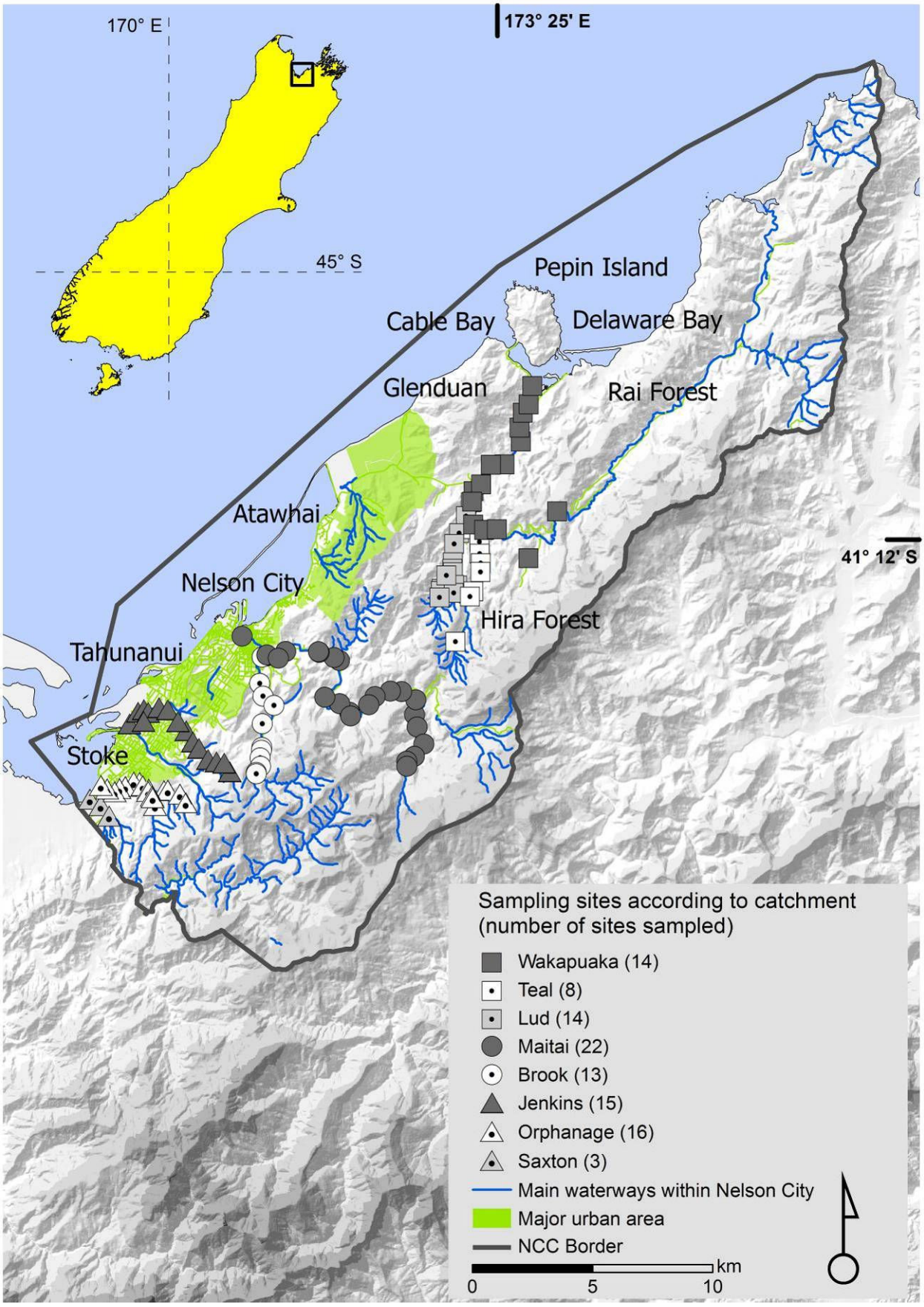


Figure 1. The 105 sites sampled within eight sub-catchments of the Nelson City Council (NCC) area between January and April 2008.

The length of streams surveyed within catchments ranged from 1.5 km (Saxton Stream) to 15.8 km (Maitai River) with stream orders from 1 (upper Wakapuaka River, upper Lud, upper Orphanage, upper Jenkins and upper Saxton Streams) to 4 (lower to middle Maitai and lower Wakapuaka Rivers). Stream order classifications were derived from the New Zealand Freshwater Fish Database (NZFFD; National Institute of Water & Atmospheric Research (NIWA)). Average stream width varied between 0.2 m (upper Lud Stream) to 27 m (lower Wakapuaka River) and average stream depth was between 0.05 m (upper Lud Stream) to 0.7 m (Teal Stream). Altitudes within catchments ranged from 0 to 350 metres above sea level (m.a.s.l.) with a maximum distance inland from the sea of 18.3 km.

Data analysed for the NCC area were supplemented with records from the NZFFD, a database that contains information on the distribution of New Zealand's freshwater fish (National Institute of Water & Atmospheric Research (NIWA)). I used records collected between 1998 and 2008, as these were the most current and accurate. To be able to compare my NCC records with the data obtained from the NZFFD, I also reduced the database search criteria to an altitude range from 0 – 350 m.a.s.l. and a distance inland range from 0-18.3 km inland. These criteria represent the physical habitat conditions used for my sampling in the NCC area and observations were restricted to those using similar sampling methods.

1.2.1.2. The upper South Island

To evaluate the status of Nelson City's freshwater fish distribution, richness and community composition, I compared the NCC area patterns with those from its surrounding geographic region. The area analysed covered the entire north coast of the South Island, including the Nelson, Tasman and Marlborough regions (Fig. 2). The upper South Island (USI) area measures 33,322 km² (i.e., Marlborough region, 17,700 km²; Tasman region, 14,831 km² and Nelson region, less the NCC area, 791 km²) with the NCC area itself constituting just over one percent

of the USI land area (443 km²). In total, there were 455 freshwater fish records for the NCC area and 1605 records for the USI in the NZFFD between 1998 and 2008, with an altitude range of 0-350 m.a.s.l. and a distance from sea range of 0-18.3 km obtained by either electric fishing and/or night spotlighting sampling techniques. These search criteria were the same as those for the NCC area, to ensure USI data came from comparable habitat conditions (distance inland and altitude).

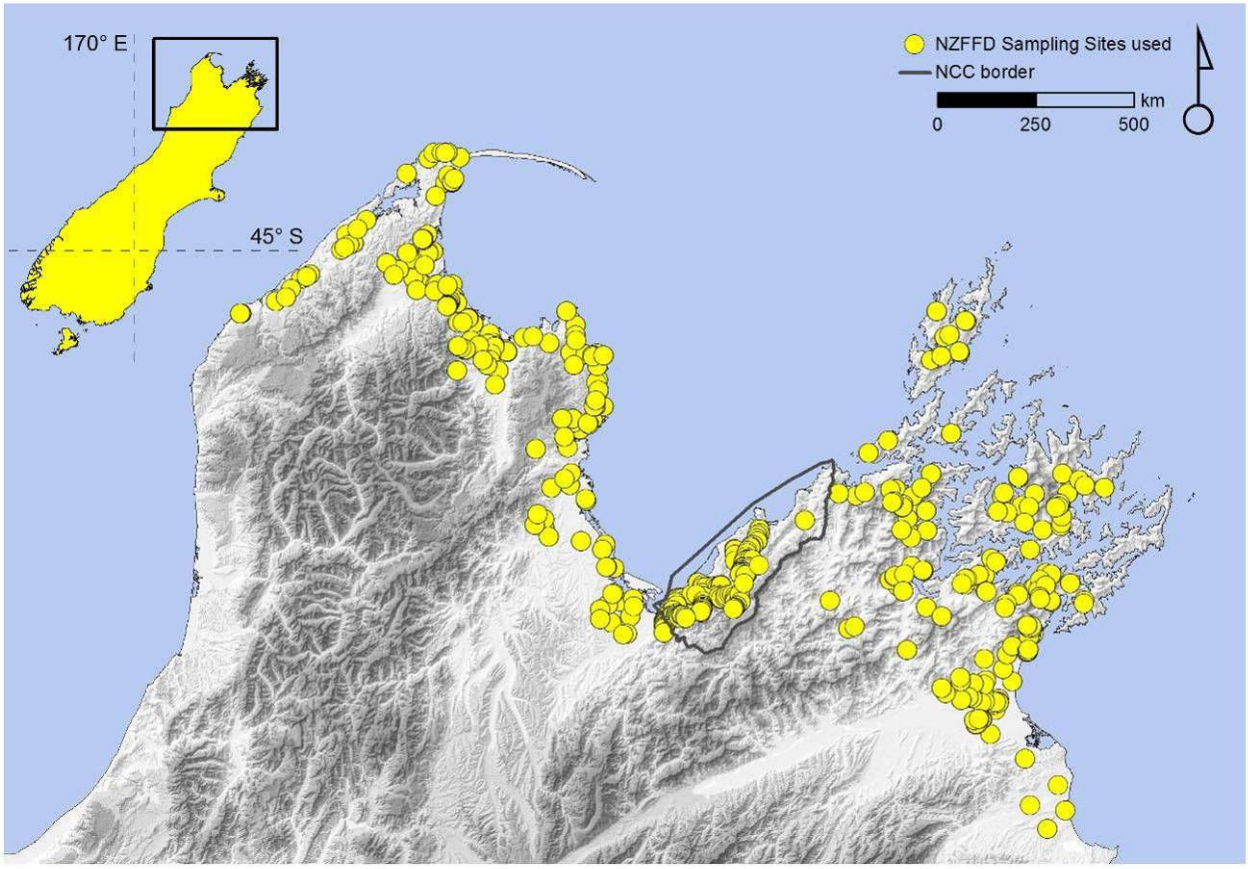


Figure 2. Sampling sites used for comparison between native fish species distribution in the upper South Island and the Nelson City Council (NCC) area within the NZFFD between 1998-2008, with a distance from sea range from 0-18.3 km, an altitude range from 0-350 m.a.s.l. and sampled by electric fishing and/or spotlighting methods.

1.2.2. Sampling

1.2.2.1. NCC area field sampling

A total of 105 stream reaches (150 m length) in eight subcatchments were sampled (Fig. 1) by spotlighting (78 %) and electric fishing (22 %) between January and April 2008. The streams were surveyed every 500 m in an upstream direction beginning at either the mouth of a stream at the sea (e.g., Maitai and Wakapuaka Rivers) or at a confluence with another stream (e.g., The Brook, Lud and Teal Streams) to the most upstream point where access was possible. On average five sites (over a 2.5 km stream length) were sampled each day. Sites surveyed by spotlighting were qualitatively sampled by searching for 30 minutes per site to locate and identify fish species. The sampling involved two people, each using a handheld spotlight powered by a 12 V battery. This method is effective for nocturnally-active species where water clarity enables all habitats to be observed (Jowett & Richardson 1994; Allibone & Caskey 2000; Young & Clark 2006). Single-pass qualitative electrofishing was also conducted by two people using a battery-powered backpack electric fishing machine (Kainga, EFM300; NIWA Instrument Systems, Christchurch, New Zealand), operated at 150–300 V depending on the water conductivity. Single pass electrofishing is suitable for assessing distribution of freshwater fish in New Zealand because all species are collected with equal probability on the first pass (Jowett & Richardson 1996).

Once fish were spotted or stunned they were caught with dip nets or stop nets, identified to species and returned to where they were caught. Most fish were able to be identified, but fish that could not be caught by spotlighting or were too immature for an accurate identification were recorded as unidentified. The position of each sampled reach was recorded with a GPS and entered into a Geographic Information System (GIS). Altitude, reach slope, maximum downstream slope and distance to sea for each sampling point were derived from a 25 m digital

resolution raster GIS layer (New Zealand South Island 25 m resolution Digital Elevation Model (DEM), estimated elevation precision = one metre) using ArcMap 9.2 (ESRI ArcView; ESRI 2006).

At each sampling site (Fig. 1) habitat data were recorded according to the NZFFD survey form. Variables measured included average stream width and depth (measured with a measuring tape), visually-assessed water clarity and colour (e.g., clear and uncoloured), average percentage of substrate composition (i.e., boulder [>264 mm], large cobble [128-264 mm], small cobble [64-128 mm], gravel [8-64 mm], fine gravel [2-8 mm], sand [0.06- 2 mm], or silt [< 0.06 mm]), average percentage of riparian and catchment vegetation (e.g., native, exotic, scrub, willow or swampland), average riparian and catchment land use (e.g., farmland or urban) as well as an average percentage of habitat type (e.g., pool, riffle or run). Chemical water quality characteristics were measured with a calibrated handheld YSI 556 multi-parameter system and included specific conductivity (µs/cm at 25° C), pH, dissolved oxygen (mg/l) and temperature (°C).

1.2.2.2. Upper South Island digital data sampling

Land-use data for the USI, including Nelson City, were obtained from the New Zealand Land Cover Database Version 2 (LCDB 2; Ministry for the Environment), a database that translates satellite images into information (polygon shape) on different land cover types in a GIS. The LCDB 2 is a hierarchical development of 70 land-use types that have been reduced to six for this study. These included: urban land use (e.g., dump or residential), exotic vegetation cover (e.g., gorse, broom, exotic forest), native vegetation cover (e.g., fernland, indigenous hardwoods), agricultural land use (e.g., high producing grassland or shelterbelts), estuarine and marine land use (e.g., herbaceous saline vegetation) and freshwater related “land use” (e.g., the extent of freshwater bodies (lake, ponds), herbaceous freshwater vegetation, river). With regard to the

latter the LCDB 2 only included polygons of medium to large water bodies in its land-cover information data and for streams, the amount of freshwater "land use" increased with stream size. Thus I was able to use the freshwater land use variable as a proxy for stream size in my analyses.

As urbanisation was expected to be more intense along lower and intermediate stream reaches, and because it is difficult to separate the influences that affect the upper stream sites due to a lack of sampling or inaccessibility, for instance, I focused the LCDB survey on lowland reaches (i.e., an altitude range from 0-110 m.a.s.l. and an inland penetration range from 0-11.5 km). The percentage of land-use cover for each NZFFD sampling spot within the USI was then derived from a 500 m longitudinal section along the river (i.e., 500 m upstream and 500 m downstream for each NZFFD record) with a 100 m strip on either side of the stream (Fig. 3). Moreover, to be able to compare fish distribution within the USI with fish distribution in Nelson City, any geo-topographical features that occurred predominately in the USI, but not in the NCC were excluded in the analysis. These included the main stems of the Whanganui Inlet, the Aorere River, the Takaka River, the Motueka River, the Moutere Inlet, the Waimea River, the Waimea Inlet and the Wairau River. The small tributaries of these rivers and inlets were not excluded from the search, hence NZFFD records for Jenkins and Orphanage Streams (Waimea Inlet) were still part of the analysis.

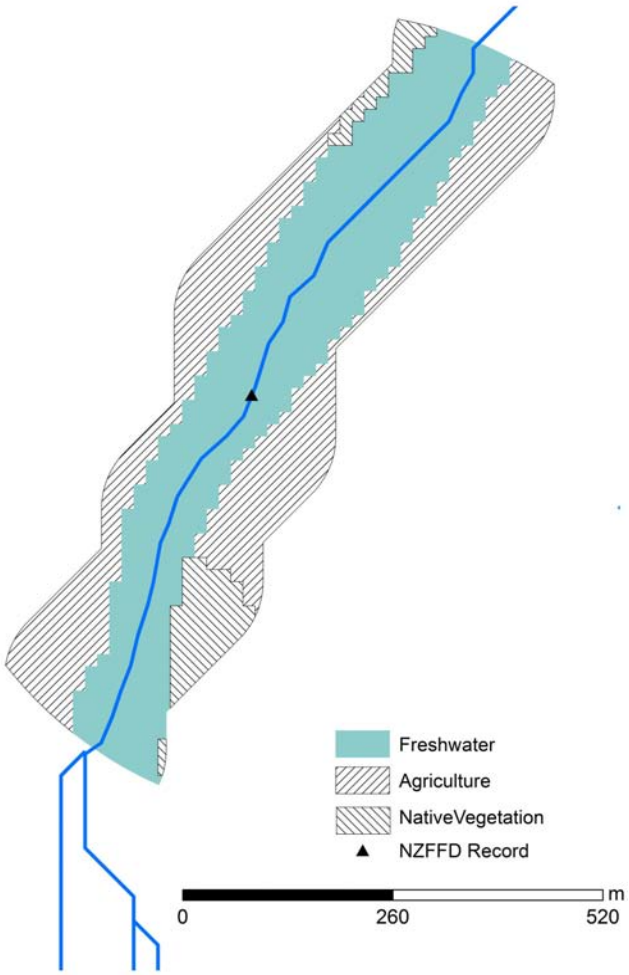


Figure 3. Riparian buffer created for the USI and NCC areas to derive the percentage of land use (from LCDB 2) for a 500 m upstream and downstream strip and 100 m strip on either side for each NZFFD record on lower and intermediate stream reaches (altitude range from 0-110 m.a.s.l., inland penetration range from 0-11.5 km). Here 53 % freshwater, 39 % agriculture and 8 % native vegetation.

To investigate the effect of measurement scale on my estimates of land-use patterns, I extended the 500 m up- and downstream buffer to the entire catchment upstream (Fig. 4). This enabled me to compare the influence of different land-use types on fish distributions between a small (i.e., 500 m buffer) and a large (upper catchment) spatial area by conducting a canonical correspondence analysis (CCA). I derived the data for fish presence from the same NZFFD records as for the riparian buffer analysis (i.e., lowland stream reaches). The land-use data (i.e., percentage of native vegetation of the total upstream catchment) were provided by the

Freshwater Environments of New Zealand (FWENZ; Snelder *et al.* 2005). FWENZ is an environmental classification of the national river network which defines the full range of remaining natural habitats and ecosystems, including the upstream areal percentage of native vegetation cover for each sampling point (Snelder *et al.* 2005). As the FWENZ data are based on stream reaches (Fig. 4), I estimated the percentage of land-use cover for each sampling point on a particular stream reach based on the FWENZ dataset. This enabled me to derive the exact percentage of catchment native vegetation upstream of each NZFFD record. I expected native vegetation to be the predominant land-use type determining fish distribution within a catchment, which is why I split up the environmental variables into 'native vegetation' and 'other land cover' in the analysis. I repeated this analysis for all NZFFD records as well as my original field sample locations (i.e., NCC).

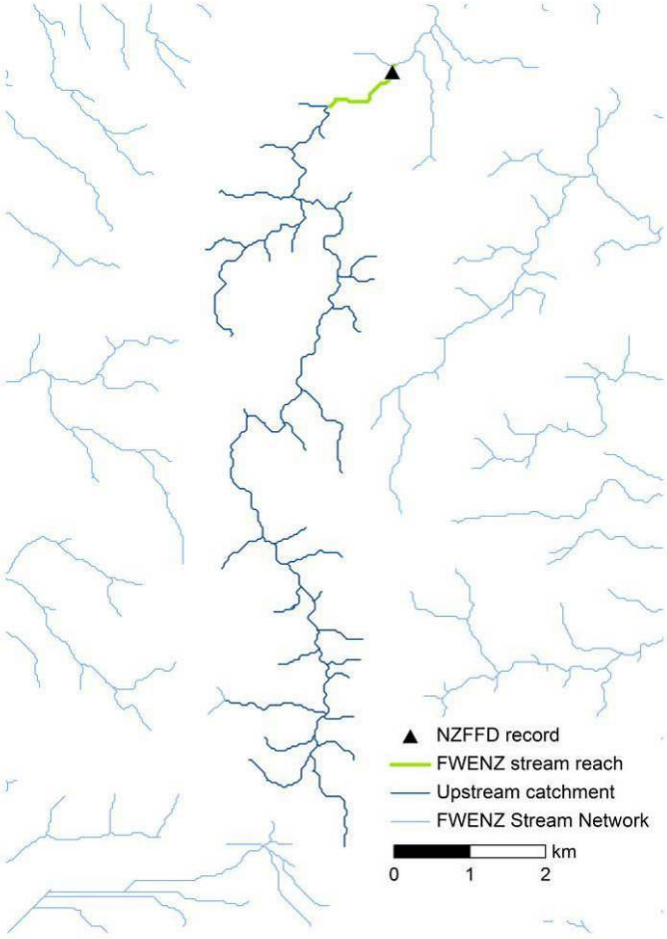


Figure 4. The percentage of native vegetation present upstream per catchment was derived from the Freshwater Environments of New Zealand (FWENZ) classification network at each NZFFD record. This helped determine whether fish distribution was influenced by catchment-wide land use.

To be able to assess the impact of culverts on fish distribution within the NCC area, their abundance, geo-topographic position (e.g., distance inland and elevation) and type (e.g., perched, pipe or box) were derived from a GIS, and associated with each sample point (from my original NCC and the NZFFD site records). This database included comprehensive information about bridges, culverts, dams, fords as well as stormwater drains and sewer pipes. Analogous culvert data were unavailable for USI NZFFD sites. Finally, the steepest downstream slope, distance inland, and elevation values for each sampling spot within the NCC and the USI were derived

from a 25 m resolution raster GIS layer (DEM) with an estimated one metre precision of the elevation layer (ESRI 2006).

1.2.2.3. Statistical data analyses

To investigate the association between fish distributions, environmental variables (e.g., chemical and physical) and land-use type, I conducted a direct gradient analysis (canonical correspondence analysis, CCA) in PC-ORD version 4.01. (McCune & Grace 2002). This multivariate analysis enables site-to-site patterns in community structure to be assessed in relation to environmental factors. Sites that have similar composition of species will tend to have similar scores for an axis. I used CCA, firstly, to study fish community composition and its association with chemical and physical variables in the NCC area, and secondly, to investigate the association of fish communities and land-use cover in the USI, including the NCC area. The latter was then used to compare fish communities and their relationship to urban land use between an impaired area (i.e., Nelson City) and an unimpaired area (i.e., USI). Fish species richness patterns may differ due to different land-use types and intensification. To investigate this hypothesis, patterns in species richness were compared between regions using chi-square tests. Specifically, I compared the proportion of sites with one, two, three, four, five or more fish species per site between the USI and the NCC area. To be able to test whether inland penetration differed between two common migratory species (inanga, *Galaxias maculatus*) and banded kokopu, *Galaxias brevipinnis*), I conducted an Analysis of Variance (ANOVA) to compare mean inland penetration between species and areas.

Using my original NCC dataset only, I further investigated relationships between native fish species richness and factors associated with urbanisation using an information-theoretic model-selection approach (Burnham & Anderson 2002). Specifically, I used this approach to identify possible combinations of anthropogenic (e.g., downstream culvert abundance) and

biogeographic (i.e., inland distance) factors that most influenced fish distribution in the NCC area. Independent variables included the subset of measured factors most likely to be affected by urbanisation (i.e., number of downstream culverts, conductivity (ln[x] transformed), proportion of fine gravel (arcsin transformed), and the presence/absence of non-native salmonids), plus inland distance (ln[x+1] transformed) as a biogeographic control. Predictor variables included distance inland (ln[x+1] transformed), number of downstream culverts, conductivity (ln[x] transformed), proportion of fine gravel (arcsin transformed) and the presence/absence of non-native Salmonids, with the total number of native species found per site (ln[x+1] transformed) as the response variable. My *a priori* set of candidate models encompassed 14 hypotheses about the effects of in-stream conditions and/or inland distance on local fish species richness. All models were ranked using Akaike's Information Criterion corrected for small sample size (AIC_c) and evaluated with respect to ΔAIC_c, the change in AIC_c relative to the best model (Burnham & Anderson 2002). I then gauged the relative support for each model, i, relative to the top model using Akaike weights [$w_i = e^{(\Delta AICi / 2)}/\Sigma e^{(\Delta AICi / 2)}$] and evidence ratios ($w_{\text{top model}}/w_i$), and plotted the data for visual representation in a separate graph.

All statistical analyses were conducted with STATISTICA 8.0 unless otherwise stated.

1.3. RESULTS

1.3.1. Freshwater fish species distribution and inland penetration in urban Nelson

A total of 455 fish species records including my 105 sampling sites for the NCC area were available between 1998 and 2008, showing 16 freshwater fish species plus the native crustacean koura (*Paranephrops planifrons*, Table 1). Of these 16, 14 are native and eleven are migratory. Longfin eel (*Anguilla dieffenbachii*), koura and giant kokopu (*Galaxias argenteus*) are listed as threatened (Hitchmough *et al.* 2007). However, while the first two species are relatively common in the area, the latter only has two records (less than one percent of the total number of sites sampled in the area) in the database (Table 1). There are no records for shortjaw kokopu (*Galaxias postvectis*) and none for non-migratory galaxiids such as dwarf galaxias (*Galaxias divergens*) in the NCC area, although both species are present in other parts of the USI (e.g., Golden Bay and the Marlborough Sounds). Non-native fish species in the area included widespread brown trout (*Salmo trutta*) and one record for rainbow trout (*Oncorhynchus mykiss*). Fish species present per site decreased with altitude (Fig. 5) as more fish species occurred at sea level than at upland sites. Five to ten species were found at low elevation sites, whereas no more than four species occurred at sites at or above 50 m in elevation.

Table 1. Fish species (including koura) sampled with either electric fishing or night spotlighting within the NCC area from the NZFFD between 1998 and 2008, with a distance from sea range of 0 - 18.3 km, and an altitude range from 0-350 m.a.s.l. Species are ranked from most to least numerous records. Threat categories were derived from the New Zealand Threat Classification System lists 2005 (Hitchmough *et al.* 2007). Introduced species are indicated by *.

Scientific name	Common name	Type of diadromy	Threat category	Records in NCC	% of occurrence	Mean Distance Inland (km)
Anguilla dieffenbachii	longfin eel	catadromous	gradual decline	130	29	6.9
Paranephrops planifrons	koura	none	gradual decline	83	18	6.3
Salmo trutta *	brown trout	none	not listed	47	10	8.4
Galaxias maculatus	inanga	catadromous	not listed	40	9	1.8
Gobiomorphus cotidianus	common bully	amphidromous	not listed	32	7	4.7
Anguilla australis	shortfin eel	catadromous	not listed	31	7	2.6
Gobiomorphus breviceps	upland bully	none	not listed	24	5	8.1
Galaxias fasciatus	banded kokopu	amphidromous	not listed	19	4	4.0
Gobiomorphus huttoni	redfin bully	amphidromous	not listed	15	3	3.8
Galaxias brevipinnis	koaro	amphidromous	not listed	11	2	9.3
Retropinna retropinna	common smelt	anadromous	not listed	9	2	4.8
Gobiomorphus gobioides	giant bully	amphidromous	not listed	6	1	2.0
Cheimarrhichthys fosteri	torrentfish	amphidromous	not listed	5	1	1.6
Aldrichetta forsteri	yelloweye mullet	marine wanderer	not listed	3	<1	0.3
Grahamina nigripenne	estuarine triplefin	marine wanderer	not listed	3	<1	0.3
Galaxias argenteus	giant kokopu	amphidromous	gradual decline	2	<1	2.1
Oncorhynchus mykiss *	rainbow trout	none	not listed	1	<1	10.8
TOTAL	17			455	100	

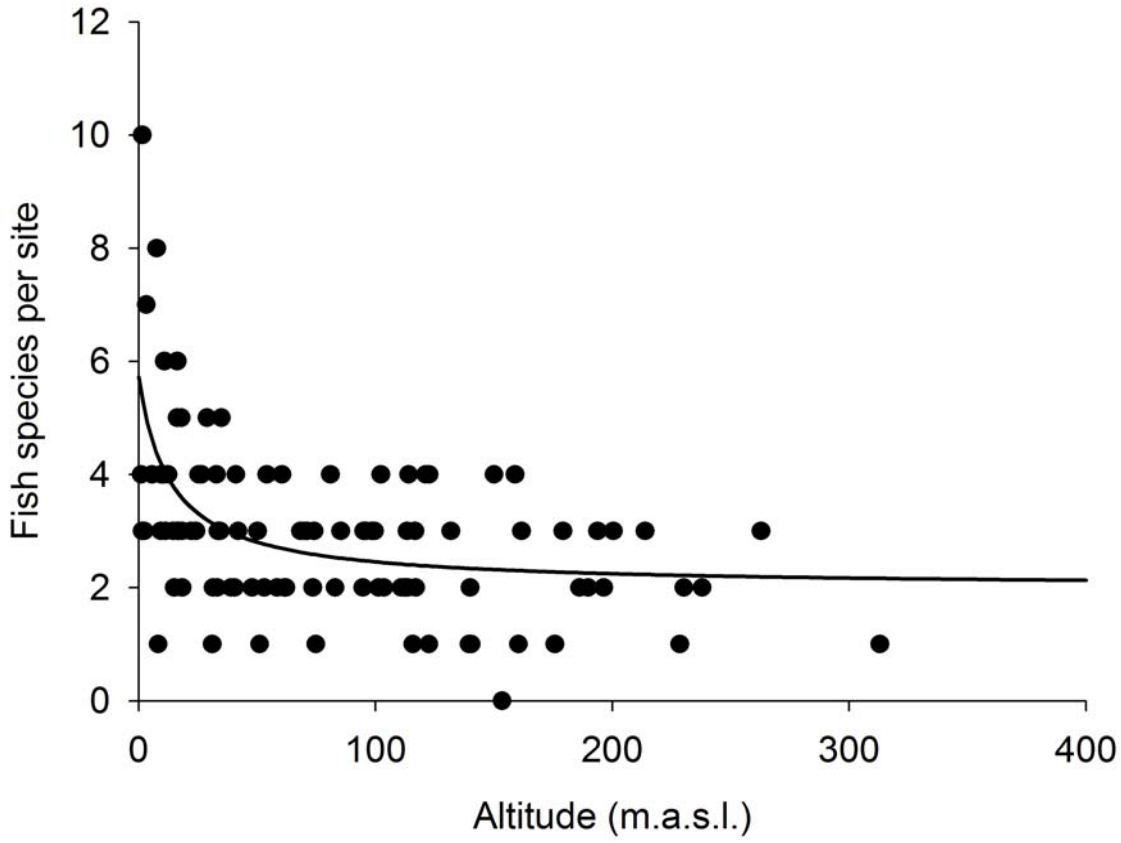
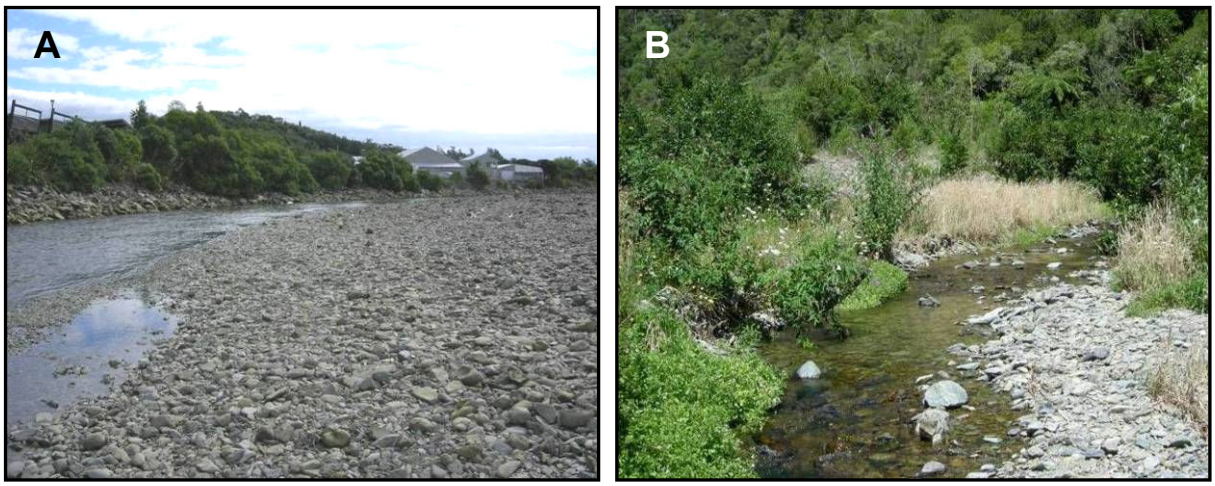


Figure 5. Number of fish species present per site along an altitude gradient from 0-350 m.a.s.l., sampled between January and April 2008. Fitted non-linear regression; $R^2 = 0.29$, df = 3, $P < 0.001$.

Estuarine triplefin, yelloweye mullet and common smelt were found in the lower, sea-water influenced, reaches of rivers such as the Wakapuaka and Maitai and the Jenkins and Saxton Streams (Photograph 2A, Table 1).



Photograph 2. The tidal-influenced lower reaches of the Maitai River (A) provide suitable habitat conditions for estuarine triplefin, yelloweye mullet and common smelt with slow flowing water velocities, whereas the middle reaches of The Brook Stream (B) have a combination of high velocity and pool habitats and medium to large boulders, creating suitable habitat for koaro.

Inanga and giant kokopu occupied the lower reaches of Nelson's waterways, banded kokopu the middle and koaro the upper (Photograph 2B, Fig. 6A). Longfin eel was the most ubiquitous species found within the NCC area, occupying entire rivers from lower to upper stream reaches, whereas shortfin eel were less common and mostly present in the small coastal streams (i.e., Jenkins, Poorman Valley and York Streams) (Fig. 6B).

Bullies were found throughout the NCC area, with giant bully recorded in lower reaches and upland, redfin and common bullies along entire river lengths (Fig. 6C). Upland and redfin bully were found at high elevations (i.e., 230 m.a.s.l.) and distances far inland (i.e., 17.7 km) in the Maitai River, however, redfin bully was absent in The Brook Stream, a tributary of the Maitai River, potentially indicating barrier issues (Fig. 6C). Rainbow trout was only recorded once in the middle reaches of the Teal River, whereas brown trout was much more widespread in the area (Fig. 6D). Although brown trout had the third highest number of occurrences in the NZFFD, it only occurrs in ten percent of records within the NCC area (Table 1).

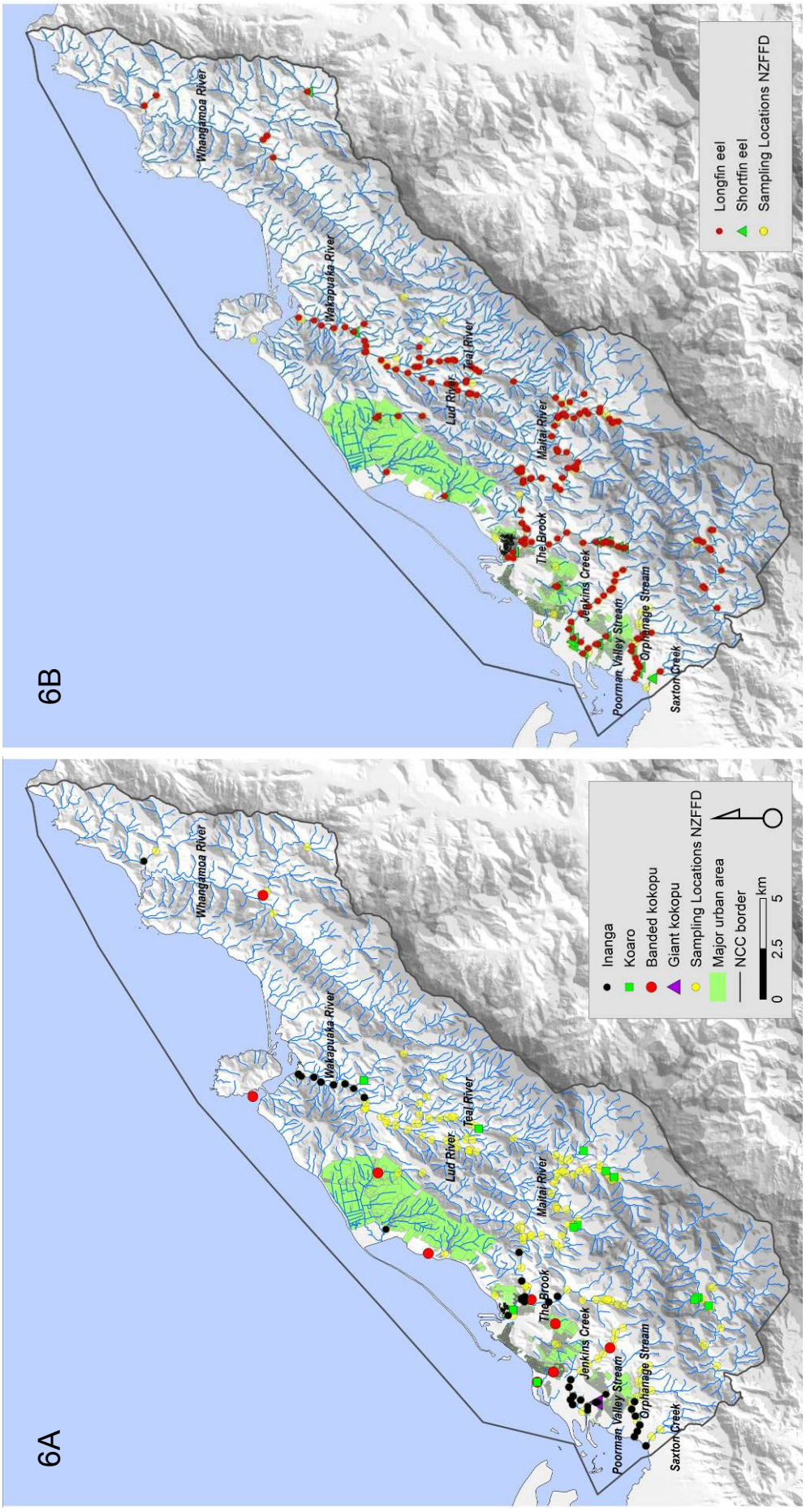


Figure 6 A & B. Individual fish species distribution (NZFFD record from 1998-2008) within the NCC area grouped by family; A = Galaxiidae, B = Anguillidae.

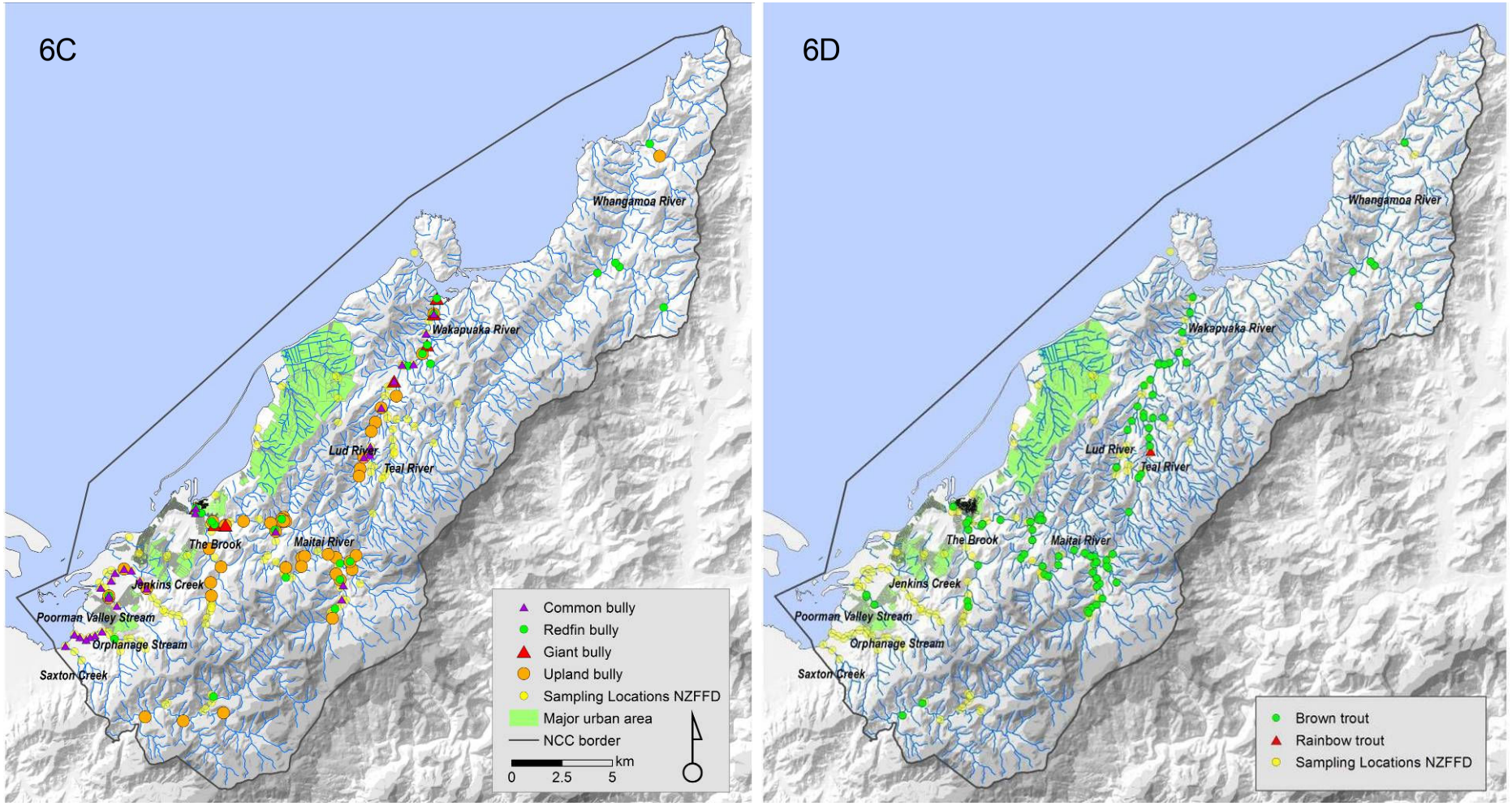


Figure 6 C & D. Individual fish species distribution (NZFFD record from 1998-2008) within the NCC area grouped by family; C = Gobiidae and D = Salmonidae.

1.3.2. Community composition

1.3.2.1. Community composition within the Nelson City Council area

On axis one of the CCA ordination (Fig. 7A) downstream reaches tended to be on the right-hand side, middle reaches had intermediate values and upland stream reaches were on the left. Thus changes in fish community structure associated with axis one were linked to the position of the site in a catchment.

The bi-plots of physico-chemical variables and species (Fig. 7B) show characteristics generally typical for upland streams (i.e., high altitude, large distance to sea, fast running water and undercut banks) and were associated with the left-hand side of axis one. In comparison, physico-chemical variables generally typical for lowland stream reaches (i.e., increased temperature and conductivity, backwater and sandy and muddy substrate) were linked to the right-hand side of axis one. Axis two describes stream size characteristics with high stream order, high mean flow and high mean width associated with low axis two scores.

Species location on the diagram with respect to the axes indicates their association with an axis (and therefore their presence in a community; Fig. 7B). Hence, koaro and banded kokopu, for example, occur at upstream sites of the upper Wakapuaka River and upper Jenkins Stream (Fig. 7A). Fish species occurring on the right-hand side of axis one (e.g., inanga, common bully) can be put into a second group where sites have similar physico-chemical characteristics (e.g., increased temperature and conductivity), such as the lower Jenkins Stream or lower Maitai River (Fig. 7A). Species that occur along the middle of axis one (i.e., koura, longfin eel) have less specific relationships with physico-chemical habitat variables and are widespread throughout the NCC area.

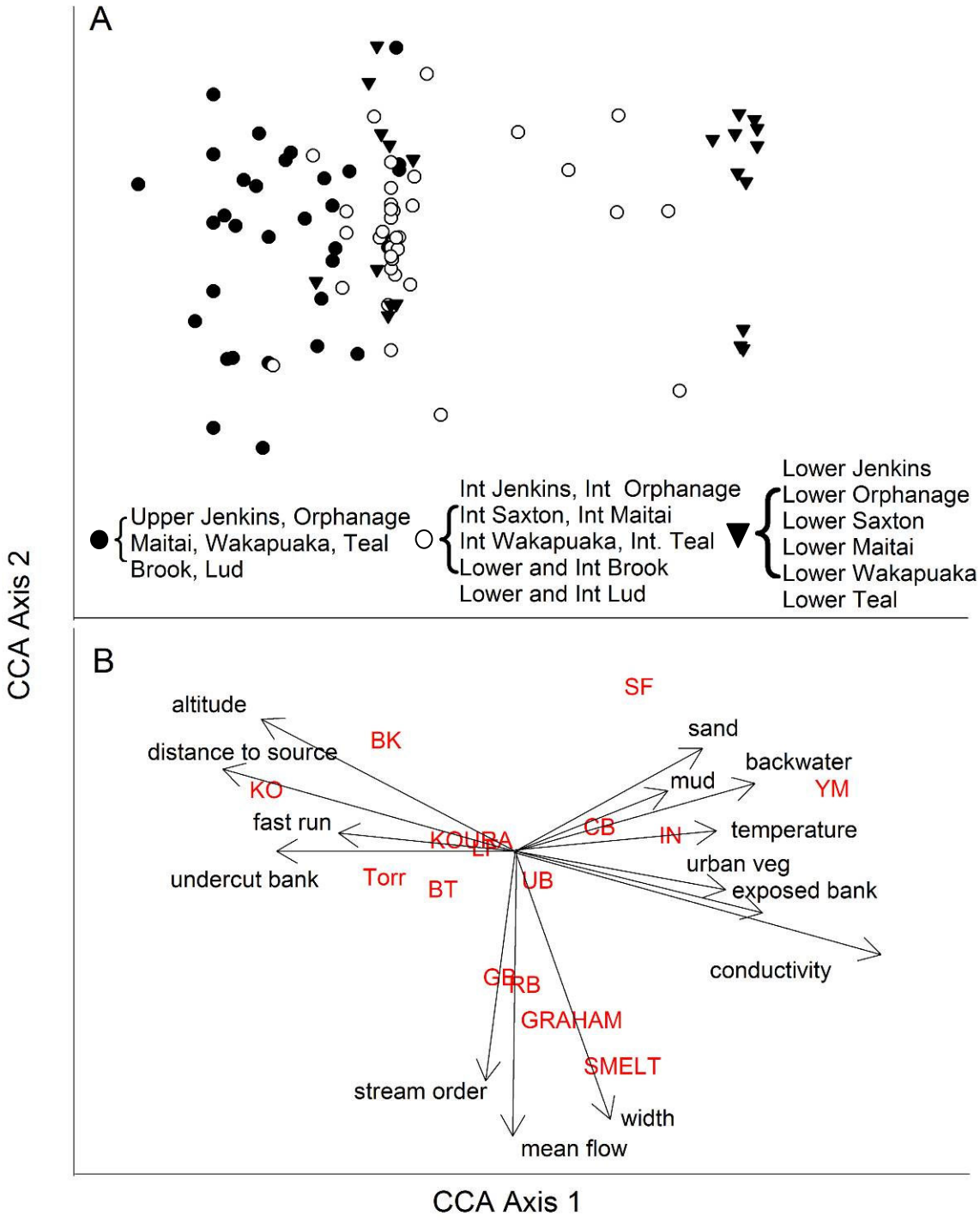


Figure 7. A canonical correspondence (CCA) ordination showing (A) sites and (B) species with associated environmental vectors for the Nelson City Council sample area. The site scores of streams sampled (A) show lowland sites on the right-hand side of axis one, middle stream reaches in the middle of axis one, and upland sites sampled on the left-hand site of CCA axis one. Bi-plots of the freshwater fish scores on the first two canonical axes (B) indicate the influence of environmental variables and individual species on these community patterns; BK=banded kokopu, BT=brown trout, CB=common bully, GB=giant bully, GRAHAM=cockabully, IN=inanga, KO=koaro, KOURA= freshwater crayfish, LF=longfin eel, RB=redfin bully, SF=shortfin eel, Torr=torrentfish, UB=upland bully, YM=yelloweye mullet.

1.3.2.2. Comparison of community composition between the Nelson City Council area and the USI

Site scores of the Northern South Island (i.e., USI including the NCC area) in the CCA ordination showed a trend for NCC sites to be on the right-hand side, and USI sites on the left-hand side of CCA axis one (Fig. 8). Thus fish community structure was again linked to the position of a site in the region. In total there were 209 sites from the NZFFD (lower and intermediate stream reaches) and 22 fish species (including koura and Paratya) recorded in the USI dataset, including Nelson City (Fig. 8A). The bi-plots of environmental variables quantified using the 500 m riparian buffer approach indicated land cover types with high impacts (i.e., urban and agriculture) and species more tolerant to intensive land use (e.g., eels) were associated with the right-hand side of axis one. In comparison, less intensive land-use types (i.e., native vegetation) and species susceptible to poor habitat quality (e.g., koaro, shortjaw kokopu, dwarf galaxias), fell on the left-hand side of axis one (Fig. 8A). CCA axis two was linked to stream size with small water bodies on the lower part and large water bodies on the upper part of the axis. Stream size can also be related to distance inland with waterways close to the sea on the top, and streams further inland on the bottom, of axis two. Accordingly, estuarine species (e.g., cockabully, yelloweye mullet) occurred at the top of axis two and species generally found in intermediate stream reaches (e.g., dwarf galaxias), occurred at the bottom of CCA axis two.

A separate CCA analysis using the same site and species scores as in Fig. 8A (i.e., the Northern South Island) derived from the land-cover entire catchment approach showed similar associations with the environmental bi-plots (Fig. 8B). Hence, sensitive fish species associated with native vegetation (e.g., dwarf galaxiid, shortjaw kokopu) occurred mostly at USI sites (left-hand side of axis one, Fig. 8B). Species that occurred on the right-hand side of axis one were located at sites within the NCC area (e.g., common bully, longfin eel, Fig. 8B). The latter were therefore most tolerant to land cover types such as agriculture or urban land use. Similar to Fig.

8A, axis two of the catchment-scale CCA describes stream size with communities associating with large water bodies in lowland sites occurring at the top (e.g., smelt) and communities associating with smaller stream sizes located in intermediate to upper stream reaches occurring on the bottom part of axis two (e.g., banded kokopu, shortjaw kokopu).

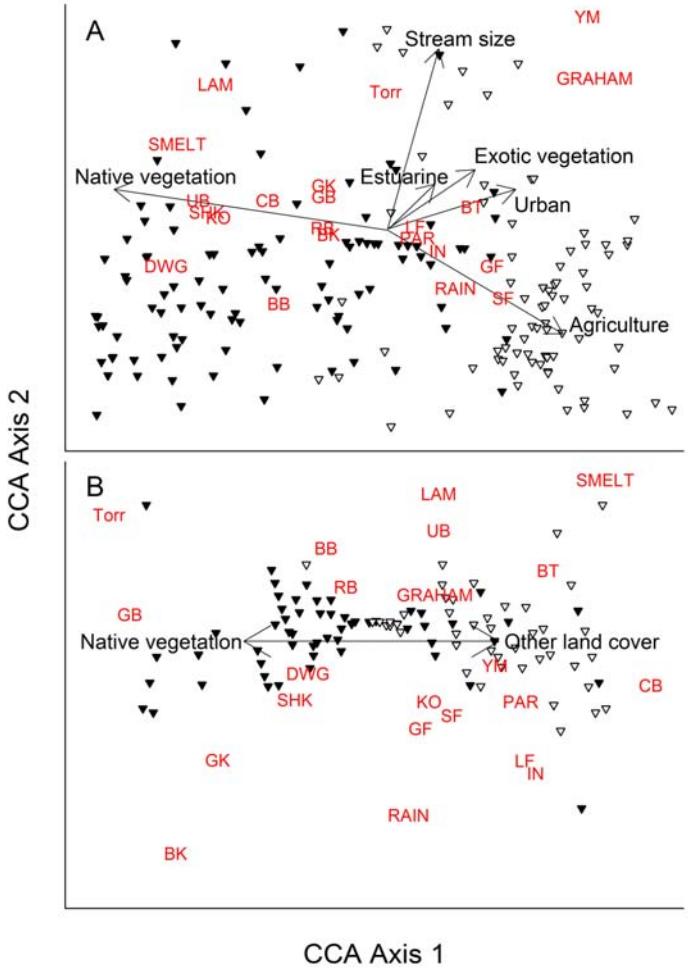


Figure 8. Plots of canonical correspondence ordination showing sites and species with associated environmental bi-plots for (A) a riparian strip buffer (500 m up-and downstream, 100 m either side) and (B) for the entire upstream catchment for each NZFFD record. The site scores show NCC sites on the right-hand side of CCA axis one (▽) and USI sites on the left-hand side (▼). Land-use bi-plots of the freshwater fish scores on the first two canonical axes indicate the influence of land-use variables and individual species on the community patterns. The arrows represent the correlation of (A) five land-use types (estuarine, agricultural, urban, native vegetation, exotic vegetation) and one habitat variable (stream size) and of (B) two land-use types (native vegetation, other land cover) with the two canonical axes. The position of each fish species represents each individual species' relative association with the ordination axis; BB=bluegill bully, BK=banded kokopu, BT=brown trout, CB=common bully, DWG=dwarf galaxias, GB=giant bully, GF=goldfish, GK=giant kokopu, GRAHAM=cockabully, IN=inanga, KO=koaro, LAM=lamprey, LF=longfin eel, PAR=freshwater shrimp, RAIN=rainbow trout, RB=redfin bully, SF=shortfin eel, SHK=shortjaw kokopu, SMELT=common smelt, Torr=Torrentfish, UB=upland bully, YM=Yelloweye mullet.

Beyond community composition, a comparison of fish species richness patterns between NCC area (Fig. 9A) and USI sites (Fig. 9B) showed a higher proportion of sites with two, three and four species per site in the NCC area than in the USI (Fig. 9). In contrast, the USI had a higher proportion of single-species and "species-rich" sites (i.e., 5+ species) than the NCC area. Based on a chi-square test of independence, these richness patterns were significantly different (χ^2 = 95.2, df = 5, $P < 0.001$).

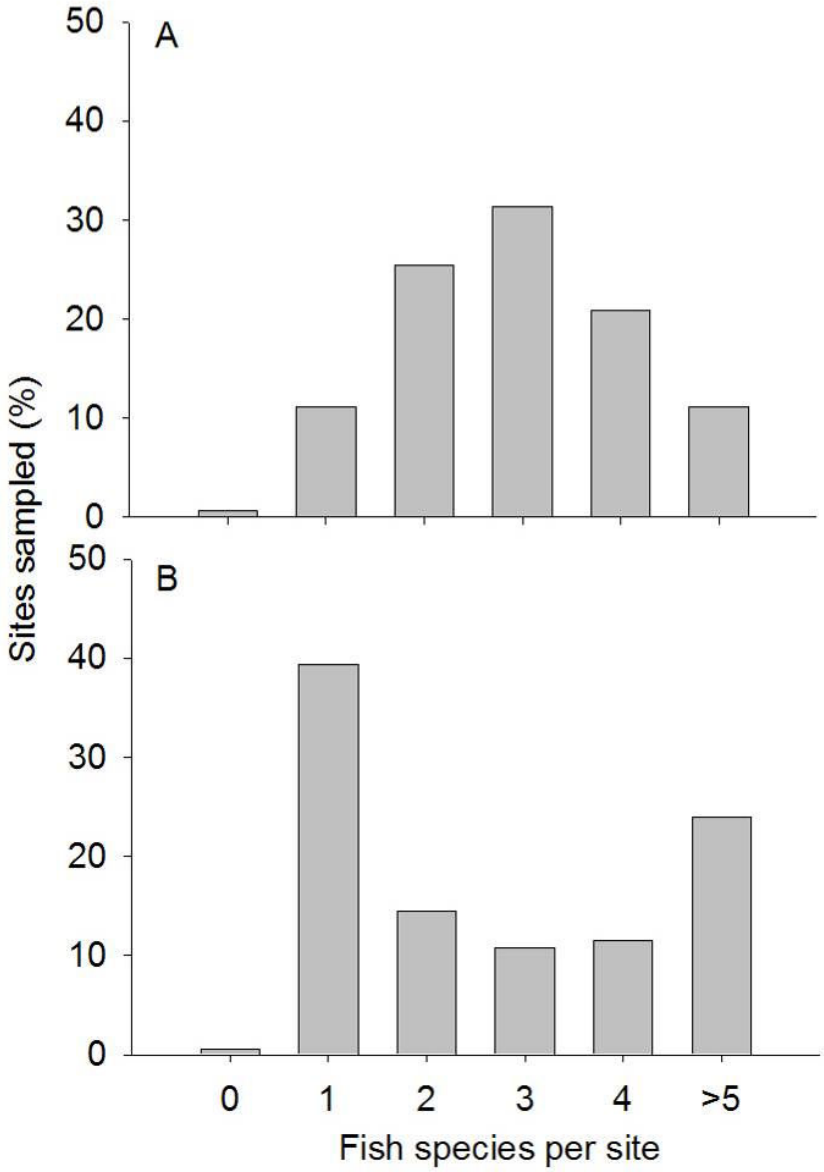


Figure 9. Relative abundance of sites in different fish species richness categories for (A) the NCC area and (B) the USI (excluding the NCC) from the NZFFD with the standard search criteria.

1.3.3. The effects of urbanisation on fish species distribution

To determine whether the fish distribution patterns found for the NCC area differed from the less intense land use of the rest of the USI, the inland penetrations for inanga (*Galaxias maculatus*; Fig. 10A,C) and banded kokopu (*Galaxias fasciatus*; Fig. 10B,D) were compared. Both species decreased in abundance with distance inland (Fig. 10). Inland penetration differed between the two areas, with both species penetrating significantly further inland in the USI than in the NCC area ($F_{2,3} = 11.08$, $P = 0.04$). The maximum inland penetrations for inanga were 2.2 km within the NCC area (Fig. 10A), and 17.1 km for the USI (Fig. 10C). Banded kokopu were recorded up to 7.2 km inland within the NCC area and 18.1 km in the USI (Fig. 10B).

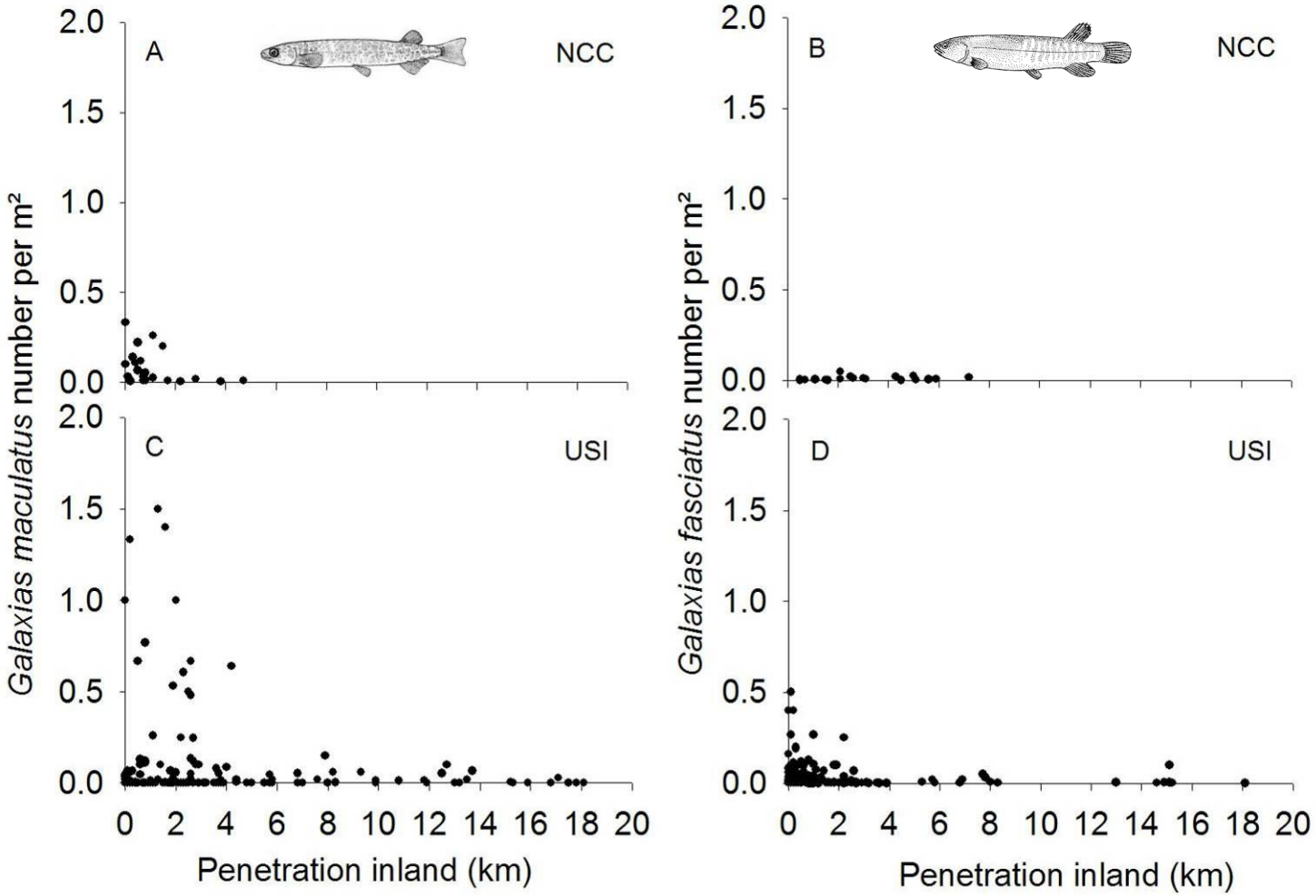


Figure 10. Abundance with respect to distance inland between the Nelson City Council (NCC) area and the upper South Island (USI) for inanga (*Galaxias maculatus*; A,C) and banded kokopu (*Galaxias fasciatus*; B,D). Data were derived from the New Zealand Freshwater Fish Database (NZFFD) with the standard search criteria.

As expected, based on the NCC richness-altitude relationship (Fig. 5), results from model-selection (AIC) indicated distance inland had an important influence on native fish species richness in the NCC area (Table 2). For example, the best model (i.e., lowest ΔAIC) among the six lacking an inland distance effect had only half the support of the worst of seven models that included distance. Overall, however, the top models also included other abiotic and biotic variables, so there was a clear effect of local-scale environmental conditions on fish species richness. Three models which included combinations of water quality, non-native species, and barrier (i.e., culvert) effects (i.e., in addition to a distance effect) were equally supported (i.e., $\Delta AIC \leq 2$, with evidence ratios near 1) for the top-model position. Compared to the influences covered by the 'Water Quality model' (i.e., non-point source pollution, sedimentation and non-native trout), I considered the impacts included in the 'Access Model' (i.e., culverts) more manageable for the NCC to address in the short term. I therefore decided to further examine the bivariate relationship between culverts and fish species richness, after controlling for the overriding inland distance/altitude effect. Fewer species than expected were found at sites with many downstream barriers than at sites with uninterrupted access (Fig. 11).

Table 2. Results of the information-theoretic model approach evaluating the influence of biotic and abiotic factors influencing fish distribution in the NCC area; *n* = 105.

Distance incl./excl.	Hypothesis Name	Model	*k*	AIC_c	ΔAIC_c	w_i	Evidence Ratio
Y	Water Quality	RICH = DIST + COND + SUB	4	-246.827	0	0.557	1
	Water Quality and Non-native Trout	RICH = DIST + COND + SUB + TROUT	5	-244.671	2	0.189	3
	Access and Water Quality	RICH = DIST + CULV + COND + SUB	5	-244.638	2	0.186	3
Y	Everything	RICH = DIST + CULV + COND + SUB + TROUT	6	-242.462	4	0.063	9
	Biogeography only	RICH = DIST	2	-235.413	11	0.002	301
	Non-native Trout	RICH = DIST + TROUT	3	-235.262	12	0.002	325
	Access & Biogeog.	RICH = DIST + CULV	3	-233.356	13	0.001	842
	Access and Non-native Trout	RICH = DIST + CULV + TROUT	4	-233.142	14	0.001	937
N	Access and Water Quality	RICH = CULV + COND + SUB	4	-216.507	30	0.000	3836641
	Everything	RICH = CULV + COND + SUB + TROUT	5	-215.337	31	0.000	6886471
	Access	RICH = CULV	2	-215.114	32	0.000	7701060
	Water Quality	RICH = COND + SUB	3	-214.505	32	0.000	10439065
	Access and Non-native Trout	RICH = CULV + TROUT	3	-213.112	34	0.000	20949935
	Non-native Trout	RICH = TROUT	2	-207.149	40	0.000	413074491

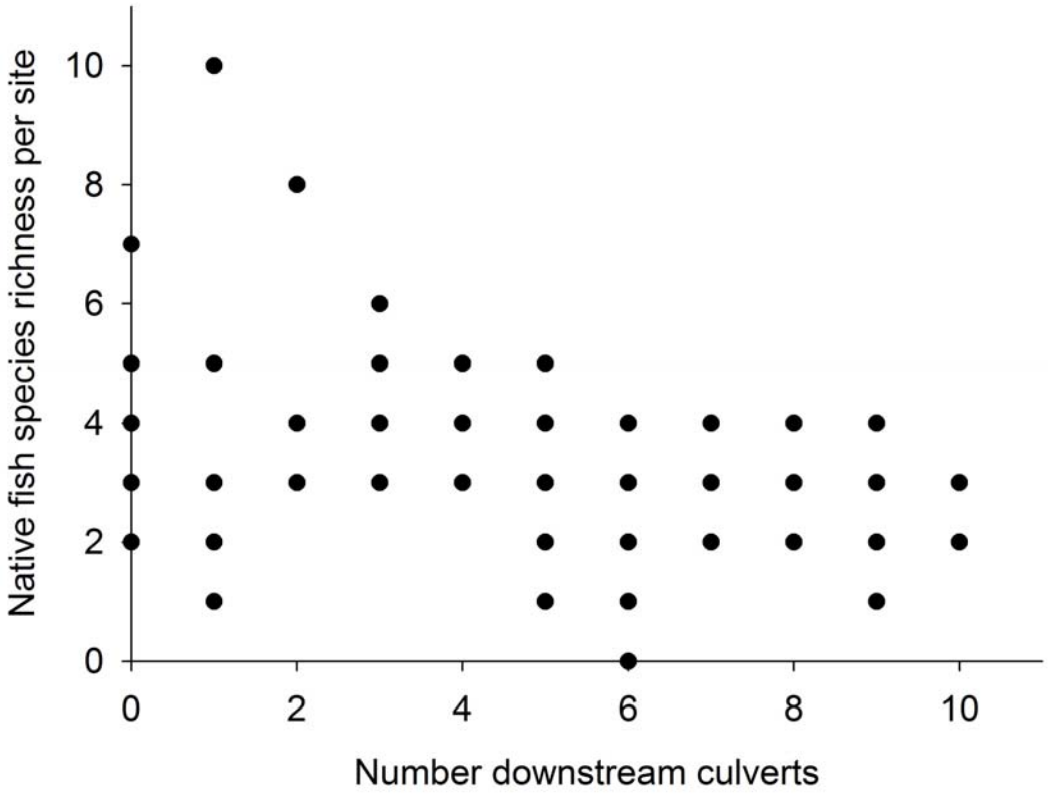


Figure 11. Fish species richness per site derived from the NZFFD in comparison to the number of culverts downstream for the NCC dataset (*n* = 105).

1.4. DISCUSSION

1.4.1. Factors influencing fish occurrence in an urban environment

New Zealand's cities are small in area, relatively young, and have small populations by world standards. Nevertheless, many streams have become greatly affected by the rapid expansion of urban development and frequently present poor habitat quality for fish and invertebrate communities (Suren & Elliott 2004). There were 17 freshwater fish species recorded in total in the NCC area, including the native crustacean koura. This is higher than the fish species richness found in Invercargill (1 fish species found), Dunedin (13), Hamilton (15) and Wellington (16), but lower than Christchurch (21) and Auckland (24) (Suren & Elliott 2004). This indicates that although urban land use might impact fish distribution in the area, there is potential to maintain high species richness in Nelson City and urban areas in general.

Individual fish distribution indicated koura and longfin eel, both species currently listed as being in 'gradual decline', were widely distributed in the NCC area (Hitchmough *et al.* 2007), potentially indicating suitable habitat conditions for these species. In contrast, other threatened fish species, such as dwarf galaxias (*Galaxias divergens*) and shortjaw kokopu (*Galaxias postvectis*), were not recorded. My data showed that both species were relatively common in the rest of the USI, but it is not clear if urbanisation is the main cause for their absence in the NCC area. Possible reasons for their absence could include an increased abundance of physical (i.e., dams, weirs) and chemical migration barriers (i.e., stormwater runoff, industrial waste), as shortjaw kokopu is one of the five whitebait species which require access to and from the sea to complete their life cycle (McDowall & Eldon 1980).

With regard to chemical migration barriers, the AIC analyses revealed water quality was the top driver determining fish distribution within the NCC area, likely causing the absence of dwarf galaxias and shortjaw kokopu within the area. Future research could focus on the effects

of increased sedimentation in urban waterways on the presence of these two species (Jowett *et al.* 1996). Moreover, Nelson City's lowland streams represent highly modified environments as almost all of the once extensive native lowland forests have been cleared (Nagashima *et al.* 2003). My analysis of the USI land-cover patterns indicated shortjaw kokopu occurred only at sites with more than 92% native vegetation cover. This is likely due to their reliance on spawning habitat with intact riparian vegetation (McDowall 1997; Charteris *et al.* 2003), conditions that are missing in the lower stream reaches of Nelson's urban streams. Similarly, the absence of dwarf galaxias could be related to deforestation of native forest in the NCC area, as my analysis showed its presence is strongly associated with sites with more than 98% native vegetation cover in the USI. However, non-migratory galaxiids are generally found in stream headwaters with long distances inland and high altitudes (Rowe *et al.* 1999), areas excluded in the standard NZFFD search for this study.

Urbanisation may also have affected the distribution of redfin bullies, a species predominantly found at high elevations and long distances inland in the Maitai River in the NCC area. Despite its commonness in the Maitai River, the species was absent in The Brook Stream, a tributary of the Maitai River. The Brook Stream is a highly modified waterway in the centre of Nelson City with an 800 m long artificial concrete channel in the lower reaches which could potentially hinder the upstream movement of redfin bullies due to, for instance, uniform water velocities (Baker & Boubée 2006). Similarly, inanga absence in the intermediate reaches, and shortjaw kokopu and banded kokopu absence in the upper reaches, of The Brook Stream could also be due to altered physical (e.g., increased velocities) and chemical (e.g., storm water drainage) water characteristics in the concrete channel. Such conditions are likely to filter out weak swimmers (i.e., inanga) and water quality-sensitive (i.e., banded kokopu, shortjaw kokopu) fish species (Baker & Montgomery 2001; Allibone *et al.* 2003).

Fish distribution may not only be influenced by habitat characteristics, but also by factors such as predation and spatial segregation due to inter-specific interactions between exotic

salmonids and native fish (Minns 1990; Harvey 1991; Townsend & Crowl 1991). Native fish communities can be profoundly impacted by the presence of introduced species (McIntosh *et al.* 1992) as invasive species, independent of their phylum, are known to be competitively dominant, profiting from already impacted habitats such as rural or urban environments (Lozon & MacIsaac 1997; Kennard *et al.* 2005; Marchetti *et al.* 2006). Trout typically exclude native fish species by either competition for food, space or predation, resulting in major declines (McDowall 2006). Such interactions are likely to influence the distribution patterns observed in the NCC area. The AIC modelling indicated the presence of trout had the second strongest effect on native fish distribution. Brown trout were generally widespread throughout the NCC area, however, they were absent in small coastal streams (i.e., Jenkins, Orphanage and Saxton Streams). In comparison to streams with trout present, which generally contained a low diversity of native fish, these streams had healthy populations (i.e., all life stages) of native fish species such as banded kokopu. Moreover, trout presence may also influence fish community composition of the entire Northern South Island (including Nelson City) as communities seemed to split into two groups. One group contained New Zealand native fish species predominantly occurring in the USI (i.e., galaxiids) and the other contained introduced species (i.e., salmonidae, goldfish), occurring in the NCC area. Trout presence might explain some of the native fish distribution patterns observed (McDowall 2006) and future research should analyse this relationship further, especially focussing on the effect of trout on migratory galaxiids in urban streams.

However, native-exotic fish interactions were only one factor likely to be driving fish distributions in Nelson City, and my analyses revealed important influences of urban land use.

1.4.2. Comparison of fish species richness and community composition between an urban-impaired and un-impaired area

Contrasting Nelson City with the adjacent region enabled me to directly evaluate differences in fish species richness between an urban-impacted area and an area with less intensive urban land use. Although the number of fish species declines inland in New Zealand (Jowett & Richardson 1996), my study indicated that sensitive species are still part of the coastal fauna in unimpaired lowland stream habitats found in the USI. This resulted in significantly different patterns in species richness in both areas. The USI had a higher proportion of sites with five or more fish species compared to the NCC area. This could be due to the USI having more sites close to the coast with good habitat conditions for fish survival. The lower species richness in the NCC area could be because fish species that generally occurred along entire river lengths in the USI, were not represented in the lower reaches of Nelson City's streams. This again is likely to be due to unsuitable habitat conditions. Common smelt, for instance, a species that prefers habitat close to the sea (McDowall 2001), co-occurred in Nelson City with species that have similar habitat requirements (e.g., cockabully, giant bully). However, in the USI, common smelt co-occurred with species normally found upstream (i.e., koaro, shortjaw kokopu), indicating that sensitive species normally were part of the coastal fish fauna, provided that habitat characteristics were suitable (i.e., good water quality, sufficient riparian vegetation).

Fish species in Nelson City included sensitive species that were relatively susceptible to low habitat quality (e.g., banded kokopu; Dean & Richardson 1999; Baker & Smith 2007) as well as fish species more tolerant to poor habitat conditions (e.g., longfin eel, inanga and common bully; Hanchet 1990; Richardson & Jowett 2002). Tolerant fish species mainly occurred in the lower reaches of urban streams, whereas sensitive fish species were mainly found in upstream reaches. Such clear-cut habitat selection by fish species between upland and lowland sites is likely due to increased urbanisation pressure on coastal sites forcing sensitive species out

of lowland sites (Marchetti *et al.* 2006). This leads to an almost entire absence of sensitive fish species in lowland sites within Nelson City. In comparison, USI fish communities associated with native forest contained both sensitive species (e.g., lamprey, koaro, banded kokopu) and species that were absent from Nelson City (i.e., shortjaw kokopu, dwarf galaxiid). There were only a few USI sites associated with urban and/or agricultural land use compared to the NCC area, and fish communities at those sites with more intensive land use contained mostly tolerant species (e.g., inanga and eels; Hanchet 1990; Hicks & McCaughan 1997; Rowe *et al.* 1999; Rowe *et al.* 2000). In total these patterns indicate that urban land use is a strong determinant of fish community patterns in the Northern South Island.

Although the impact of land use on fish communities is widely recognised in New Zealand and elsewhere (Hanchet 1990; Allan 2004; Scott 2006), there have been only a few direct comparisons of land-use impacts at both site- and catchment-scales. My results showed that urbanisation not only drives fish communities in waterways immediately surrounded by a city, but it also determines fish community composition of the entire upper catchment, where urban land use occurs in the lower reaches.

My results differ somewhat from previous studies indicating freshwater fish assemblages usually become more similar at reach scales and at zoogeographic provinces, but less similar within one region (Marchetti *et al.* 2001). In my study fish communities associated with either native vegetation or other land-use types were quite similar between site-scale and catchment-scale analyses within the USI region. The Northern South Island was regarded as a "region" in my study, but Marchetti *et al.* (2001) defined regions as "zoogeographic provinces". Regardless of the definitions, the similarity of the results from analyses at different scales is striking, and likely indicates urbanisation is a very strong driver of fish community structure.

Urbanisation affects fish distribution by multiple mechanisms, including deforestation, chemical migration barriers and the presence of non-native fish species, and further analyses

showed that migration barriers played a particularly important role for fish distribution in urban environments.

1.4.3. The effects of migration barriers on fish species distribution

Diadromous species enter rivers by migrating upstream from the sea, resulting in more species at low elevations and near the coast (Hayes *et al.* 1989; Joy *et al.* 2000). Despite fish species distribution in the NCC area following this "natural" pattern, my analyses showed that urbanisation was likely to alter inland distribution of some fish species. Inland penetration of inanga, for instance, was significantly reduced in Nelson City than in the USI, resulting in inanga being absent in stream reaches intermediate distances from the coast, such as in The Brook Stream. Inanga is considered to be a weak swimmer (Mitchell 1989; Nikora *et al.* 2003), making it vulnerable to the effects of migration barriers. However, my analyses revealed similar patterns for banded kokopu which was recorded at distances significantly further inland in the USI than in Nelson City. Although banded kokopu is considered to be a good swimmer (Mitchell 1989), and is normally distributed in middle to upstream reaches (Baker & Smith 2007), the species was still absent in many intermediate stream reaches in the NCC area, such as in The Brook Stream. The fact that two fish species with different habitat requirements and swimming behaviours were present at intermediate and upper distances inland in the USI, but absent from equivalent sites in Nelson City, likely indicates a common driver for these patterns.

The AIC modelling indicated water quality was a main determinant influencing fish species distribution within the NCC area, but migration barriers associated with culverts had an almost equal impact (i.e., the second strongest AIC model). Culverts are a man-made, common and visible feature in urban environments, and compared to the effects of water quality and trout presence on fish distribution (the other two factors in the leading models), the impacts of culverts might be more manageable in the short term.

Further analyses focussing on the effects of culverts as a driver of fish species distribution showed there was a significant decrease in fish abundance with an increase in the number of culverts downstream, potentially explaining the absence of inanga in the middle reaches, and of banded kokopu in the upper reaches, of Nelson Cities streams. By 2008, the NCC had 270 culverts recorded within the NCC area, but their actual barrier potential (i.e., whether a culvert is perched or not) for fish species is unknown. My results indicate they likely affect fish community patterns, filtering out fish species and reducing fish abundance in urban streams.

1.4.4. Implications for urban planners

Urbanisation likely has major effects on freshwater fish distributions and communities, impairing stream ecosystems by altering their environment (Paul & Meyer 2001; Wang *et al.* 2001), but its effects remain largely unknown. New Zealand's urban streams have been especially affected by the fast spatial growth of cities, but there have only been a few studies looking at the impacts of urbanisation on New Zealand freshwater fish communities (Suren & Elliott 2004). My study indicates GIS analyses, with easily acquired data, can enable the evaluation of land-use effects on fish communities without extensive field sampling. Such an analysis allows managers and councils to assess the magnitude of urbanisation effects on fish within their region. This should become a compulsory part of urban development. Once councils have analysed existing effects, future urban planning should include measures to mitigate the "impacts" of urbanisation. My results suggest that urban stream ecosystems require increased management effort, integrating both site- and catchment-wide approaches as well as the management of multiple mechanisms that drive freshwater fish distributions. However, it might be difficult to address all of the factors influencing fish distribution within an urban area. In this case, the focus should be set on the strongest drivers such as storm water pollution, or the most feasible solution such as removing physical migration barriers. Despite culvert assessments

becoming a standard procedure for councils, my study showed that there is still need for remediation.

Finally, it is necessary for urban planners and resource managers to realise that urban streams should be valued for more than their role as drainage systems. Ecologically healthy streams that are aesthetically pleasing provide attractive places to live and recreate (Suren & Elliott 2004). Urban streams may provide the only chance some people have to live near nature, highlighting the existing paradox that urban streams frequently become degraded during the process of urban development. The real challenge for planners and managers is therefore to minimise these adverse impacts, so that our children can enjoy the privilege of nature in cities for many years to come.

Chapter Two

FACILITATION OF UPSTREAM PASSAGE FOR A NEW ZEALAND MIGRATORY FRESHWATER FISH SPECIES

Banded kokopu (*Galaxias fasciatus*) whitebait climbing up a barrel installed next to a perched culvert in Golden Bay, New Zealand.

2.1. INTRODUCTION

The alteration of urban river systems by physical barriers (Chapter One), is common throughout the world and has often been linked with impacts on the freshwater fish fauna up- and downstream of these structures (e.g., reduced abundances; Allibone 1999; Gibson *et al.* 2005b; Boubée & Williams 2006). Culverts have been found to impede fish movement particularly. The most predominant impacts include high water velocities both downstream and within the culvert itself (Haro *et al.* 2004), and an impassable vertical undercut drop (perch) at the culvert outlet (Adams & Whyte 1990). Thus, over recent years, much research has focused on designing ways to allow fish to pass such obstacles (Gibson *et al.* 2005b; Baker & Boubée 2006). However, despite complex engineering guidelines for stream crossings, a general lack of detailed information on the swimming performance of target species and financial constraints, have commonly led to poor design and installation of fish ramps in urban waterways (Haro *et al.* 2004). This has resulted in ongoing upstream and downstream passage issues for migratory freshwater fish species (Gibson *et al.* 2005b).

For salmonids in North America or galaxiids in New Zealand, larval/juvenile accessibility to upstream and downstream reaches is crucial for species population sustainability (McDowall 1998; Coutant & Whitney 2000). Migration is undertaken for the purposes of locating suitable habitat, growth, reproduction and food acquisition and is a vital part of the life history of these fish (McDowall 2000, 2001). The New Zealand fish fauna is distinctive worldwide in that more than half of the species are diadromous (i.e., fish which migrate between the sea and freshwater; Myers 1949; McDowall 1998; Joy *et al.* 2000), so barrier-free waterways are essential in allowing longitudinal migration to the up- or downstream reaches. However, connectivity between water bodies does not just apply to diadromous fish species. Providing successful passage throughout a waterway is also important for so-called resident riverine fish species that regularly migrate considerable

distances throughout a watershed (potamodromous species) such as golden perch (*Macquaria ambigua*; Reynolds 1983; Coutant & Whitney 2000; Haro *et al.* 2004; Griffiths 2006).

Migratory fish species, especially in New Zealand, often concentrate in lowland streams to spawn (McDowall 1993) and those streams can be disproportionately affected by urbanisation (Knaepkens *et al.* 2006). In the same way that the presence of migration barriers can be directly related to urban land use, fish decline can also be linked to urbanisation (Magaud *et al.* 1997; Suren & Elliott 2004). Thus, improving upstream passage at culverts is not only crucial because many species are an important component of New Zealand's freshwater fish fauna (e.g., food supply for many birds and predatory fish species) (Jowett 2002; Rowe *et al.* 2002), but also because five of the migratory galaxiids make up a substantial part of New Zealand's whitebait fishery (Rowe *et al.* 1992). Whitebait is considered to be a luxury product (Mardones *et al.* 2008) with values up to $83 USD per kilogram at the time of writing. However, stocks have substantially declined in New Zealand and South America in the past 100 years (Rowe *et al.* 2000; Jowett 2002; Haggerty 2007; Mardones *et al.* 2008), and consequently fishery authorities have great concerns over the effects of urbanisation on fish movement, calling for a better understanding of methods to improve in-stream connectivity.

Inanga (*Galaxias maculatus* Jenyns) are the most common species in the New Zealand (and overseas) whitebait fishery (Jowett 2002; Mardones *et al.* 2008) and it is sensible therefore to focus efforts on improving passage barriers for this species. Inanga are a small (80-150 mm long) and slender, olive-coloured fish species with a translucent juvenile phase. They are widely found throughout New Zealand at low elevations, as well as in eastern and western Australia, Tasmania, South Africa and the southern-most parts of South America. Inanga are reproductively mature after one year when they migrate downstream to estuaries in autumn to spawn a few-hundred to a few-thousand eggs amongst vegetation, upstream of the tidal salt wedge. After hatching, the larvae (*c.* 7 mm long) migrate to sea to feed and grow,

before returning to freshwater as whitebait (*c.* 45-50 mm, 21-23 weeks old) (McDowall 2000). Migratory barriers are expected to have a particularly large impact on the whitebait stage of this species, and the inability of these young-of-the-year to ascend culverts may be a crucial factor limiting inanga recruitment in a river system (Stuart & Mallen-Cooper 1999; Baker 2003). Therefore, it is essential to continue to focus on the swimming behaviour and ability of juvenile inanga to negotiate obstacles in order to better understand how this species, and other migratory fish species more generally, are able to overcome anthropogenic barriers.

There has been some research on fish passage requirements of inanga (e.g., Baker & Boubée 2006), but due to a lack of knowledge of the species' life history and behaviour at barriers, the results are often not easily applicable (Jowett 2002; Baker 2003). Moreover, most studies have focused on the adult stage, yet inanga migrate as juveniles (Kemp & Williams 2008). As this species is considered to be a weak swimmer (Mitchell 1989; Baker & Boubée 2006), determining passage requirements based on their swimming ability could likely be extrapolated to other weak swimmers, such as bully species (*Gobiomorphus* spp., Gobiidae). Hence, my study of upstream passage is also applicable to non-commercially important fish species.

The swimming capability of species at barriers is a major criterion when investigating fish passage design and depends on several environmental variables (Baker & Boubée 2006; MacDonald & Davies 2007; Tudorache *et al.* 2008). The two main variables are water velocities within (barrel) and downstream of a barrier (Haro *et al.* 2004; Peake 2004; House *et al.* 2005) and/or physiological effects related to fish size, age or fatigue (Nikora *et al.* 2003). New Zealand's native fish species do not pass barriers by jumping as salmonids do (Naughton *et al.* 2007; Kemp & Williams 2008), instead they climb vertical wet rock surfaces using either surface tension (e.g., koaro, *Galaxias brevipinnis*) or use snake-like motions (e.g., Anguillidae) (McDowall 1993, 2003; Eikaas & McIntosh 2006). Fish species that cannot climb, such as inanga or bullies, must burst swim past or over high velocity areas.

Despite these morphological and behavioural adaptations (e.g., modified climbing or swimming abilities), many species are still unable to get past in-stream barriers that are perched, lack wetted margins or create sustained high velocity flows (Boubée *et al.* 1999). Inanga are particularly affected by high water velocities at culvert entrances, hindering their upstream migration (Mitchell 1989; Baker & Boubée 2006). Determining the swimming ability of weak swimming species like inanga will allow urban planners and resource managers to appropriately design under-road crossings, facilitating upstream movement for many fish species and therefore maintaining a diverse freshwater fish fauna in urban environments.

To fully appreciate the relationship between water velocities and the swimming ability of inanga, there are three dominant swimming modes that need to be understood (Nikora *et al.* 2003). Sustained swimming is an aerobic motion that can be maintained for an indefinite period, whereas burst swimming is a short, high speed anaerobic motion that cannot be maintained for prolonged periods. The intermediate of these two is prolonged swimming, which involves both aerobic and anaerobic processes of energy supply (Weihs 1974). Once the different swimming modes of a target species are known, ramps can be designed accordingly to enable fish to proceed past barriers (Peake 2004). If ramps are to be successful, fish must be able to ascend them quickly (Baker & Boubée 2006). Hence, for management applications the knowledge of fatigue plays a particularly important role (Hammer 1995), and the swimming capability of the target fish species should be considered when designing fish ramps (Hudson 1973; Peake *et al.* 1997a; Peake *et al.* 1997b; Boubée *et al.* 1999; Tudorache *et al.* 2008).

To overcome the main issues related to culverts as migration barriers, my study focused on determining the most important variables affecting fish passage. The ultimate goal of this research was to design more effective fish passages for use in urban stream design. With this in mind, my objectives were: (1) to investigate how the climbing ability of juvenile

inanga was influenced by changing the angle and the length of a fish ramp; (2) whether upstream movement and hence passing success could be improved by providing resting opportunities and (3) what effect fish size had on passing success.

2.2. MATERIALS AND METHODS

2.2.1. Experimental design

To determine how best to reduce the effect of migration barriers on juvenile inanga, I designed an artificial ramp to simulate conditions fish face at migration barriers. This allowed the effects of ramp length, ramp angle and the presence of pools as resting opportunities on the success or failure of inanga to traverse fish ramps to be tested. To measure the passing success of juvenile inanga, a three metre long ramp constructed from corflute (polypropylene) and Astroturf was set at one of three different angles (5°, 15° or 25°) with either zero pools, one pool (at 1.5 m distance) or two pools (at 1 m and at 2 m distance) (Table 1).

Table 1. The six treatments tested including three different angles, each having two pool combinations.

Angle	5°		15°		25°	
Number of Pools	0	1	0	1	1	2

The angle and pool combinations used in my experiments were considered the most suitable to study climbing behaviour of inanga, as previous research has shown this species is able to pass a 1.5 metres long ramp at 5° and 15° angle without resting opportunities. However, to successfully traverse ramps longer than 1.5 metres and steeper than 15° they need resting pools (Baker & Boubée 2006). Thus, by installing either one or two pools at 25°, I expected upstream passage to improve compared to the study of Baker & Boubée (2006).

2.2.2. Experimental apparatus and ramp hydraulics

A recirculating freshwater flow system pumped water from a lower holding tank (40 l) to an upper holding tank (20 l) connected by a ramp (Fig. 1A). To ensure upstream passage was not restricted by water depth, mean water depth in the pools (25 mm) and on the ramp (15 mm) was always kept at least twice the body depth of the largest juvenile fish tested by regulating flow conditions. The length of the ramp was 3500 mm with 500 mm immersed in water at the lower end so the actual 'climbing' length was 3000 mm. The ramp had a total width of 300 mm and was v-shaped in cross-section (Fig. 1B).

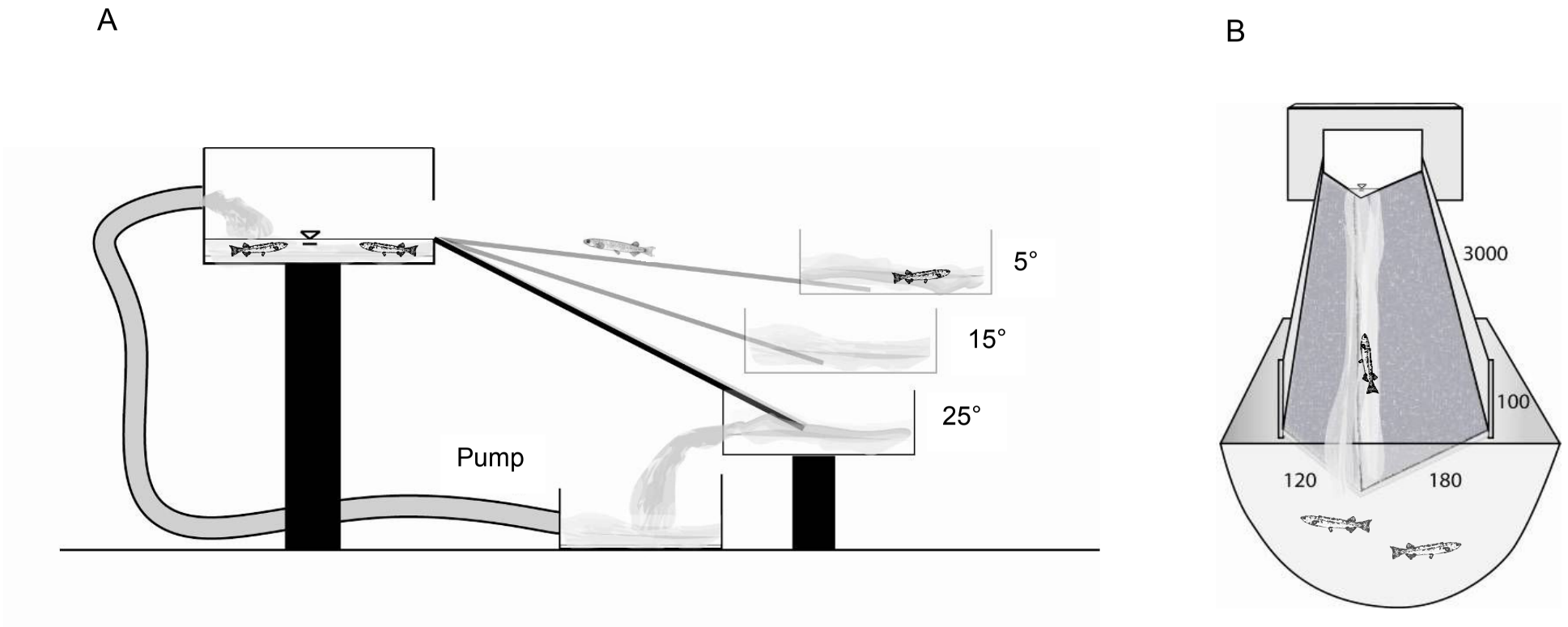


Figure 1. Schematic illustration of the ramp (A) profile, and (B) cross-section (all dimensions are in mm), showing the experimental set up.

I used Astroturf (synthetic turf) as the surface material for the ramp with a grass length of 10 mm (Photograph 1A); this material produced substratum characteristics conducive to fish climbing. The rough Astroturf surfaces also increased surface tension, enabling fish to rest (Baker & Boubée 2006). To provide further resting opportunities for fish during the climb, I created pools by deforming the Astroturf from the underside with sand bags (Photograph 1B). Pool length (200 mm) at each angle was maintained by adjusting the height of the sand bags according to the steepness of the ramp.

Photograph 1. (A) Astroturf was used to provide a rough surface allowing fish to rest during their upstream climb without getting washed back into the lower holding bin, and (B) a pool (arrow) was created at 1.5 m to provide further resting opportunities for fish.

2.2.3. Animal capture and maintenance

Juvenile (fork length < 60 mm) inanga were sourced from a small tributary of the Wakapuaka River at the Paremata Flats Reserve, within the Nelson City Council area. Fish were caught with a whitebait set-net (2 mm-mesh net) and kept for up to three days in live boxes within the stream (Photograph 2). Pilot studies have shown that holding fish in live bins reduces stress and increases passage performance (Baker 2003).

Photograph 2. A live box in a small tributary of the Wakapuaka River at the Paremata Flats Reserve enabled me to keep fish in their natural environment before being used in experiments.

A maximum of 100 fish were transported in 20 l containers to the laboratory at least 24 h before commencement of each run of the experiment to allow fish to adjust to the tank conditions (i.e., water temperature, oxygen levels) before experiments started. In the laboratory, fish were kept in an aerated 30 l tank for a maximum of three days. Fish were fed each evening with chironomids. Stream debris (i.e., rocks, algae) was added to the tank to

provide cover. The tank water was piped from a local stream close to the laboratory and replaced daily. Water temperature in the tank ranged from 13° to 19° C.

The experiments were carried out in October and November 2008 at the Aquaculture Facility of the Cawthron Institute at Glenduan, twelve kilometres northeast of Nelson City, New Zealand. Experiments were conducted between 0700 hours and 2000 hours, as whitebait generally move during daylight hours (Stancliff *et al.* 1988; Baker & Boubée 2006). For each trial, ten randomly selected fish of varying size, but all < 60 mm fork length, were placed in a lower holding tank and given three hours to pass the ramp. To minimise disturbance by observers, a curtain was erected beside the ramp after fish were placed in the lower tank. After three hours, I counted and measured (mm, fork length) the number of fish that had successfully traversed the ramps (Photograph 3). That is, fish that made it into the upper tank were successful, but fish that were still on the ramp, in the lower tank or in either of the pools after three hours were considered to have not passed the ramp. Each day up to four experiments were run (i.e., sampling day as a random factor) with each experiment having a different treatment combination. Each treatment combination was repeated five times (i.e., 5 replicates per treatment). For each replicate trial a different set of ten fish was used; fish that were used in experiments were released at the end of each day into the main stem of the Wakapuaka River, thus minimising the likelihood of recapture.

Photograph 3. After the experiment fish were measured (mm, fork length) to determine whether passing success depended on fish size.

2.2.4 Statistical analyses

Analysis of covariance (ANCOVA) was used to evaluate if pool presence (one pool) and ramp angle (5° and 15°) influenced minimum water velocity. A two-sampled *t*-test was used to test whether there was a significant difference in the number of fish passing the ramp at 15° compared to 5° slope. Fish size and the number of fish to successfully pass the ramp were square-root transformed where necessary to meet assumptions of normality and homogeneity of variances. ANCOVA, ANOVA and the *t*-test were conducted in STATISTICA 8.0.

To assess the effect of fish size on passing success at 5°, 15° and 25° ramp angles with one pool, I used a generalized mixed-effects (GME) model (using the *lmer* package in R; R Foundation 2007) with a binomial distribution (i.e., pass or fail). This allowed me to test the fixed effects of ramp angle (categorical variable) and fish size (continuous variable) on whether the fish were able to traverse the ramps while accounting for any variation that was due to running the experimental trials over different days (random block effect). I used model

simplification (Crawley 2007) with maximum likelihood (ML) estimation, where the fixed effects of ramp angle, fish size and their interaction were removed one-by-one from the model to estimate the chi-square statistic and its significance level for each of the fixed effects and their interaction on fish passing success.

I used a GME model with a normal distribution to determine whether resting opportunities (i.e., the presence or absence of one pool) influenced the number of fish (continuous response variable) that were successful in passing 5° and 15° ramp angles. Again, the main and interactive effects of ramp angle and pool presence/absence were tested using model simplification and ML (as above), after taking into account any variation due to sampling day (random block effect).

2.3. RESULTS

2.3.1. Hydrodynamics of ramp and pools

Overall, water flow on the ramp was uniform, due to the consistent substratum texture. ANCOVA indicated minimum velocities were slower in the pools than on the ramp, and increased with ramp angle ($F_{1,3} = 27.27$, $P = 0.01$, Fig. 2). However, the presence of pools did not affect water velocities on the ramp ($F_{1,3} = 7.94$, $P = 0.06$; Fig. 2). Water depth on the ramp and in pools also depended on angle, with deeper water levels on lower ramp slopes (Table 2).

Table 2. Ramp hydraulics for each of the six trials tested, and the average number of fish that successfully passed each treatment. Min. = minimum; Max. = maximum; Av. = average; R = ramp; P = pool.

Treatment (R angle/P number)	Min. Velocity R (m/s)	Max. Velocity R (m/s)	Av. depth R (mm)	Max. depth R (mm)	Av. depth P (mm)	Max. depth P (mm)	Av. water temp (°C)	Av. number passed
5°/0	0.28	0.36	19	25	-	-	15.2	3.80
5°/1	0.32	0.45	16	18	28	30	16.9	4.40
15°/0	0.34	0.47	14	16	-	-	16.6	1.00
15°/1	0.34	0.36	15	15	36	40	15.8	2.60
25°/1	0.35	0.68	17	20	29	35	15.2	0.00
25°/2	0.34	0.50	15	15	25	30	13.8	0.04

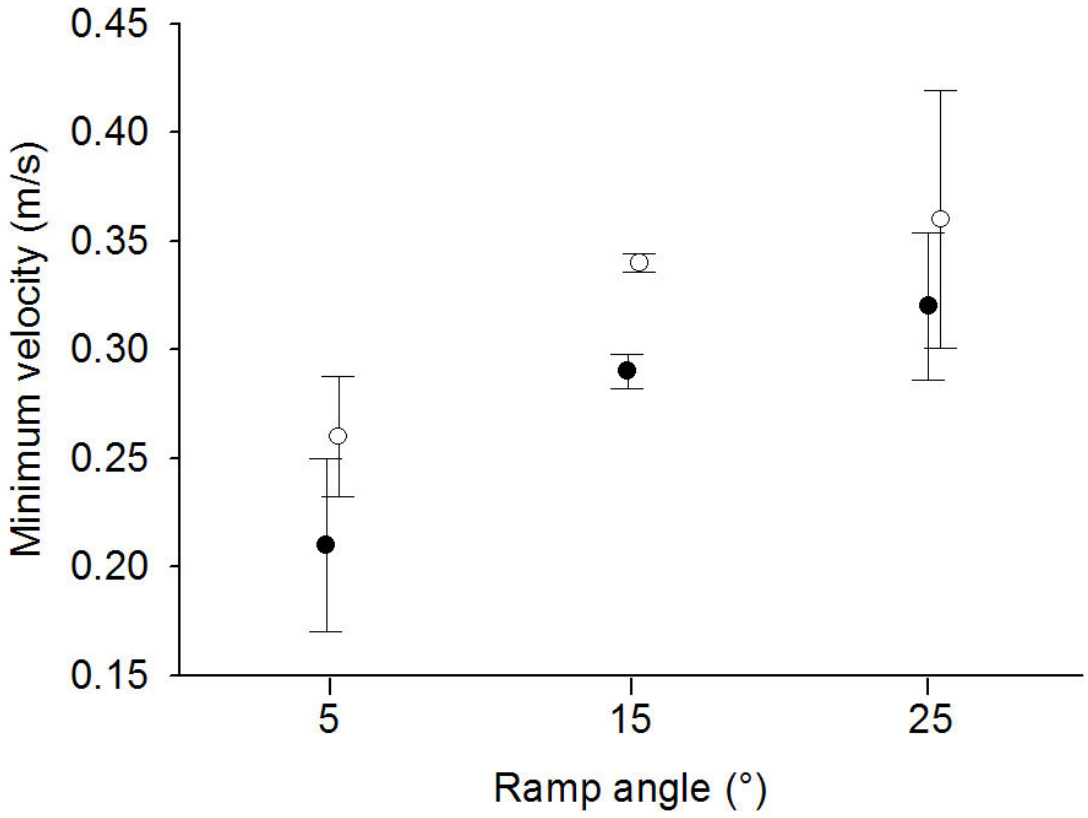
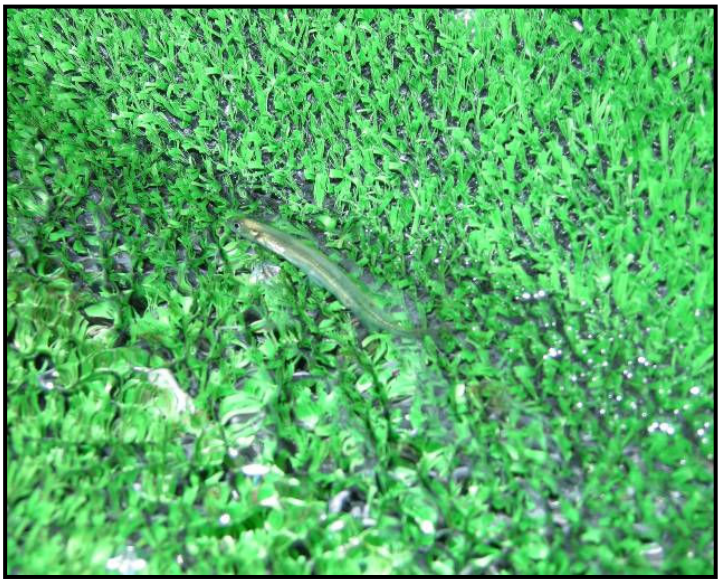


Figure 2. Mean (± S.E.) minimum velocity measured at different slopes on the ramp (◯) and in pools (●).

2.3.2. Fish behaviour

Once in the experimental channel, fish adjusted rapidly to their new environment, attempting to climb after approximately ten minutes. Fish continued to attempt upstream passage during the entire three hours of a trial. To pass the ramp, fish would begin to swim in the main current at the base of the ramp before attempting an upstream climb by burst swimming. Some fish did not succeed in entering the main channel in the first instance and were washed back into the lower holding tank. Others burst swam up the main channel until they reached a pool or moved into the wetted margin at the sides of the ramp to rest before continuing their attempt (Photograph 4).



Photograph 4. An inanga on the wetted margin of the ramp. In general fish burst swam up the ramp for approximately one metre, before they moved to the wetted margin of the Astroturf.

2.3.3. The effect of angle on fish passing success

Fish were randomly selected across all trials and were variable in size, ranging from 40 mm to 58 mm. A two-sample *t*-test showed that fewer fish were able to successfully climb ramps with a 15° angle than a 5° angle, when only one pool was present ($t_8 = 2.9$, $P < 0.05$; Fig. 3). As no fish passed at 25°, I could not use ANOVA because the homogeneity of variances assumption was violated. However, the 95% confidence interval for the mean number of fish passing at 15° (2.1 - 3.1 fish passed per trial) did not include zero, and so was significantly different to the number passing at 25°.

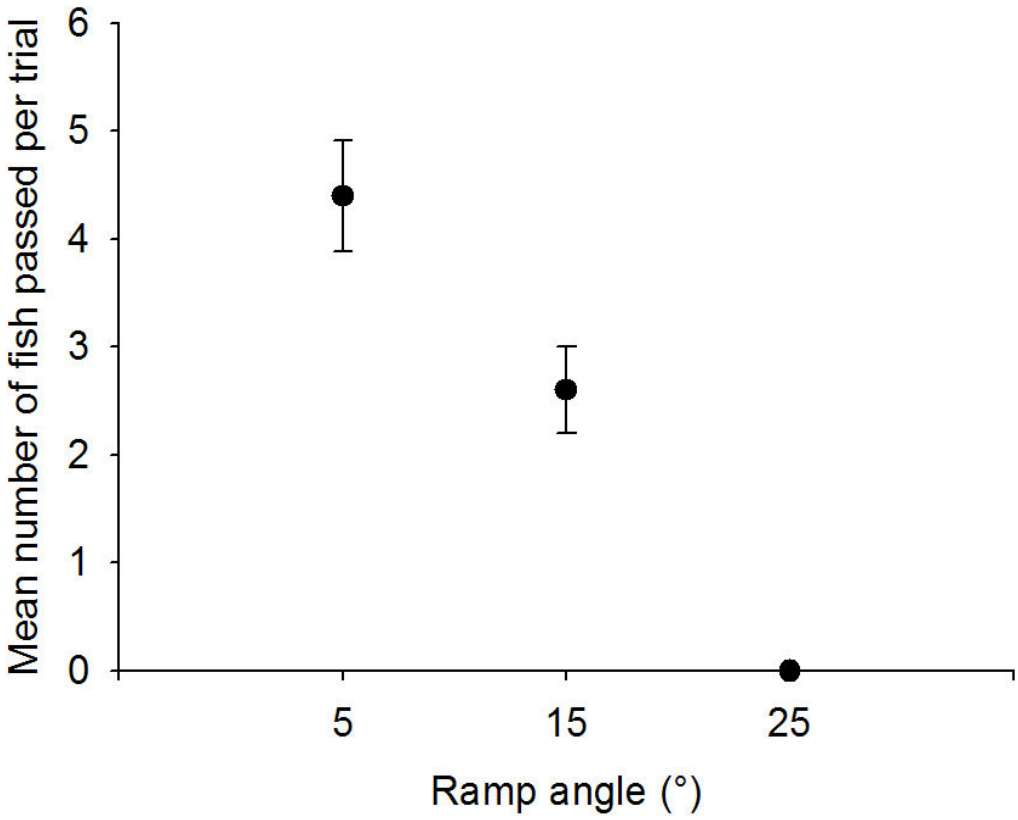


Figure 3. Mean (± S.E.) number of fish passing the ramp at 5°, 15° and 25° angles with 1 pool.

2.3.4. The effects of fish size and pool presence on passage success

A binomial mixed-effects model indicated larger fish were able to climb steeper angles (Table 3a; Fig. 4). There was no significant interaction between the two variables (Table 3a). To pass the ramp with a 50% probability, the analysis indicated fish needed to be larger than 49.6 mm at a 5° slope and larger than 54.6 mm at a 15° slope (Fig. 4).

A linear GME indicated that pool presence did not significantly influence fish passing success, but fish were more likely to successfully climb shallow ramps than steep ramps (Table 3b; Fig. 5). There was no significant interaction between the two variables (Table 3b). Although the presence of pools did not significantly improve upstream passage ($P = 0.08$, Table 3b), there was a trend for pools to increase the likelihood of successful passage, especially at 15° (Fig. 5).

Table 3. Generalised mixed-effects models testing the fixed main and interactive effects of (a) ramp angle (5°, 15°, 25°, one pool) and fish size on the success or failure of fish to climb the ramp (binomial distribution), and (b) ramp angle (5° and 15°) and pool presence (no pool or one pool) on the number of fish that successfully climbed the ramp (normal distribution), while accounting for variation due to sampling day (random block effect). Model simplification (Crawley 2007) was used to estimate the chi-square (χ^2) statistic and its significance level (P <0.05) for each of the fixed effects and their interaction. df = degrees of freedom, var. = variance due to sampling day (random block effect), P values <0.05 are indicated in bold.

Response	**Predictor**	**Var.**	χ^2	**df**	***P* value**
(a)	Sampling day	276.33			
Fish passed/failed	Angle		7.01	1	**0.029**
	Size		229.17	1	**<0.001**
	Angle x Size		5.05	2	0.079
(b)	Sampling day	0.004			
Fish passed/failed	Angle		9.57	1	**0.002**
	Pool		2.91	1	0.080
	Angle x Pool		1.71	1	0.190

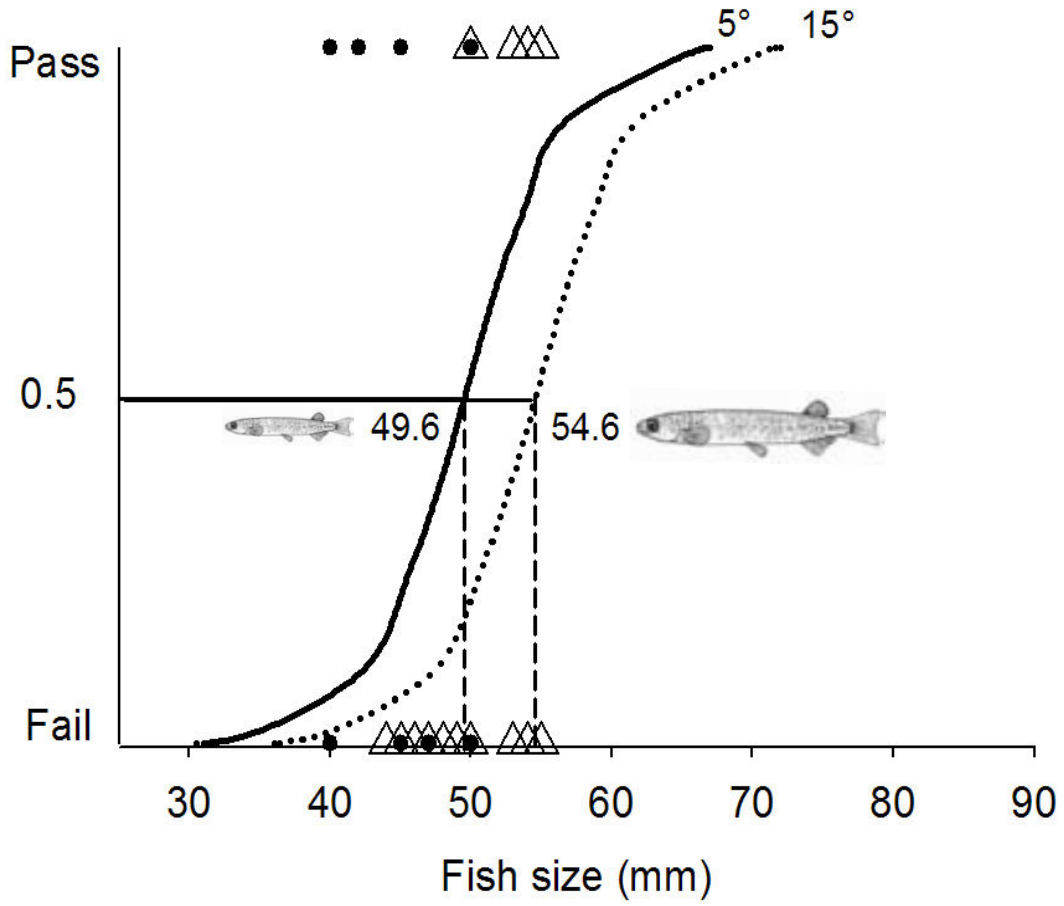


Figure 4. The size of fish that successfully climbed (pass) the ramp versus those that did not (fail) at 5° (●) and 15° (△) angle with no pools, and their predicted passing size at 5° (▬) and 15° (····) ramp angles determined by a logistic mixed-effects model.

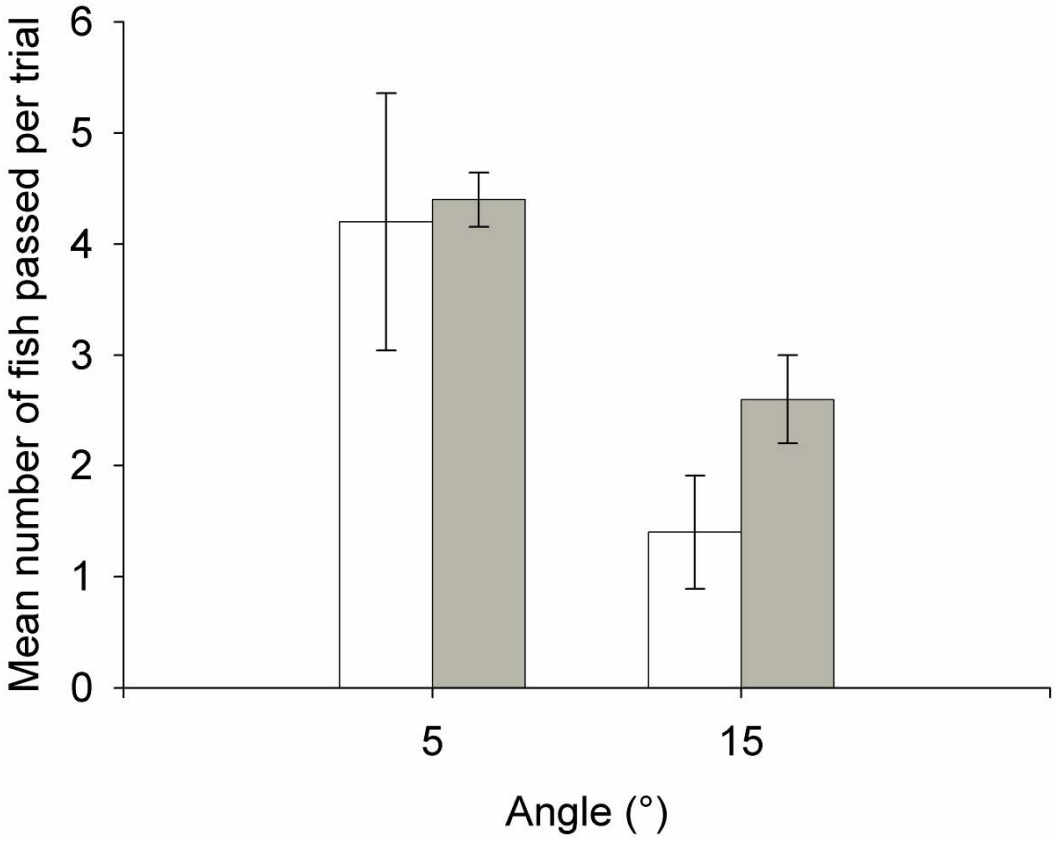


Figure 5. Mean (± S.E.) number of fish that passed the ramp at 5° and 15° with zero (▭) pools or one pool (▭), tested with a linear mixed-effects model.

2.4. DISCUSSION

The alteration of river systems by migration barriers is common throughout the world and impacts on freshwater fish faunas have often been documented (Chapter One; Fitch 1996; Allibone 1999; Paul & Meyer 2001; Pringle 2001). Hence, designing an effective fish passage for new, as well as for existing in-stream structures will play an important role in the population sustainability of many fish species (Schilt 2007). Although a variety of facilities have been designed to ease upstream fish passage (e.g., Baker & Boubée 2006; Richmond *et al.* 2007), engineers and managers mostly focus on large in-stream structures (e.g., dams and weirs), disregarding smaller, but potentially equally inhibiting, structures. It is often the small structures such as culverts (diameter < 1.5 m) that present a barrier, especially to small fish species (Stevenson *et al.* 2008). One way to mitigate these effects is the installation of fish ramps.

When studying fish ramp designs, there are several factors that need to be considered. Fish size, for example, plays an important role, as larger fish are generally able to swim faster and longer (Mitchell 1989; Harding *et al.* 1998; Peake 2004; Tudorache *et al.* 2008). However, previous studies have neither included small fish (juveniles) nor slope aspects in their evaluation of fish passage design (Boubée *et al.* 2001; Knaepkens *et al.* 2006; Naughton *et al.* 2007). Consistent with previous research on adult fish (Baker & Boubée 2006), the juveniles in my study needed to be larger to pass a steeper gradient. Specifically, fish had to be 5 mm larger to surmount the ramp at 15 degrees than those at 5 degrees. This result highlights the importance of considering size and life stage of target fish species when constructing fish ramps (Nikora *et al.* 2003). When planning fish passage at culverts during, for instance, juvenile migration periods, fish ramp angle should be as small as possible to provide successful passage for weak swimming species or life stages.

Many New Zealand indigenous fish species tend to be small (50 – 70 mm length) when beginning their upstream migration, so this size-dependent ability to climb means there is

potential for even slight changes in the physical environment to affect their ability to move through a watercourse (Baker & Boubée 2006; Stevenson *et al.* 2008). Water velocities are one of the main parameters likely to affect passage of small fish species (Baker & Boubée 2006). To successfully design fish ramps, water velocities on, within and downstream of the barrier need to be considered (Peake *et al.* 1997b; Haro *et al.* 2004; Peake 2004; House *et al.* 2005). The minimum water velocity tested in my study increased with slope, and although water velocities at 15 degrees still allowed 12% of the juvenile inanga tested to successfully negotiate the ramp, none passed at 25 degrees. Thus, angles steeper than 15 degrees could potentially increase the velocity on fish ramps beyond the ability of weak swimming fish (Baker & Boubée 2006; Tudorache *et al.* 2008).

High-velocity flow zones characterize many natural rivers and are often unavoidable or even intentional features of fishways, dams and culverts (Warren 1998; Parker 2000; Haro *et al.* 2004). For velocity to be considered a barrier, it must exceed the physiological or behavioural capabilities of the target species and life stage of the fish that are expected to be migrating (Parker 2000; Haro *et al.* 2004; Stevenson *et al.* 2008). Increased water velocities can also cause changes in burst swim distances. Boubée *et al.* (1999) showed the maximum burst swim distance for adult inanga changed from 6.2 m at 0.35 m/s to 2.1 m at 1.0 m/s. These measurements were made for adult inanga at a single channel angle. However, they illustrate how burst swim distances for juveniles could vary accordingly with fish ramp angle and hence water velocities.

As water velocities are closely related to ramp angle, the latter is another important variable that needs to be considered when designing fish ramps. Significantly fewer fish passed the ramp at 15 degrees than at 5 degrees in my study, and only one fish traversed the ramp successfully at 25 degrees (with two pools installed). Thus, fish ramps constructed at 25 degree angle at a length of three metres would only provide passage to fish species that are better swimmers than inanga, such as redfin bullies (*Gobiomorphus huttoni*; Baker & Boubée (2006)).

Fish ramps designed to allow juvenile inanga passage should therefore ideally be installed at an angle of 5 degrees and not steeper than 15 degrees.

Ramp length may also have affected fish passage in my study. My ramp length provided realistic estimates of passage ability past intermediate sized obstacles such as small diameter culverts (diameter < 1.5 metres), enabling some individuals to successfully negotiate the full length of the ramp. However, Baker & Boubée (2006) looked at juvenile inanga (44 – 55 mm) climbing ability over a 1.5 m long ramp and found that 75 % of fish successfully negotiated all ramps at a 15 degree angle, independent of surface substrate. In my study only 12 % of fish passed at an angle of 15 degrees over a ramp twice the length (3 m). Other factors influencing passage success may have differed between our studies, but inanga passage increased by six times on a ramp half the length. Thus, the trade-off between ramp length and angle deserves further investigation.

In many cases, the height of a structure determines the length of a ramp, as the higher the structure, the longer the ramp must be. Ramp length, in this case, may be a problem for fish if the distance they can travel at any one time is restricted to less than the full length of the ramp. Some fish may be able to swim faster than the velocity of water on fish ramps over short distances, but not up long ramps (Parker 2000). The high energy costs involved during the upstream movement may mean fish become exhausted before they reach the end. In these situations, resting areas are recommended (Boubée *et al.* 1999), explaining why I investigated the influence of pool presence. The lack of resting areas existing in a uniform environment, such as on bare fish ramps or in a culvert, may be problematic for fish attempting to ascend a fish ramp (Clay 1995; Rajaratnam *et al.* 1997). Uniform velocities are usually not found in natural rivers, as riverbed substrata break up water flow, resulting in diverse flow environments (Stevenson *et al.* 2008). Flow non-uniformity provides resting areas of low velocity, enabling fish to move upstream without exceeding their fatigue limit (House *et al.* 2005). Moreover, preferential selection of low velocity refugia as resting opportunities is well known in fast flow environments (Nikora *et al.*

2003; MacDonald & Davies 2007). Hence, offering pools to rest during their upstream climb may improve passage success for any given fish species (Dominy 1973; Rajaratnam *et al.* 1997; Kim 2001). Although the presence of pools had no significant effect on the number of juvenile fish that passed the ramp at any angle in my study, the results were very close to being significant for the pool presence by angle interaction. Fish may not have been exhausted enough after a one and a half metre climb at any slope to actually use the pools to rest in my study. Furthermore, pools that have been installed at fish ramps formed turbulences (macro-eddies) which can hinder fish resting ability (Kim 2001; Liao *et al.* 2003b; Richmond *et al.* 2007; Pavlov *et al.* 2008). Further investigation of the role of pools as resting opportunities for juvenile fish would therefore be worthwhile.

Suitable surface substrate may also offer sufficient resting opportunities for fish during their climb and likely affected passage in my study. Surface substrata in general should be as rough as possible to both reduce flow and widen the wetted margin, offering fish opportunities to rest (Hegberg *et al.* 2001b; Haro *et al.* 2004). Miradrain© (raised 24 mm high cups on a flat surface, each 16 mm apart) produced the most successful passage in the study by Baker & Boubée (2006). However, I tested Miradrain© as surface material prior to my Astroturf experiments, and found it less suitable as fish were unable to maintain a grip on the material. Until further data have been gathered on the effects of resting opportunities, fish ramp designs should provide at least one wetted margin with rough surface substrata to enable fish to rest during their upstream movement.

2.5. CONCLUSION

Several techniques have been employed to resolve the problem of how to design effective fish ramps for culverts, but often constraints of time, effort and money have limited the scope of the experiments to find suitable and applicable solutions to migration barriers (Castro-Santos *et al.*

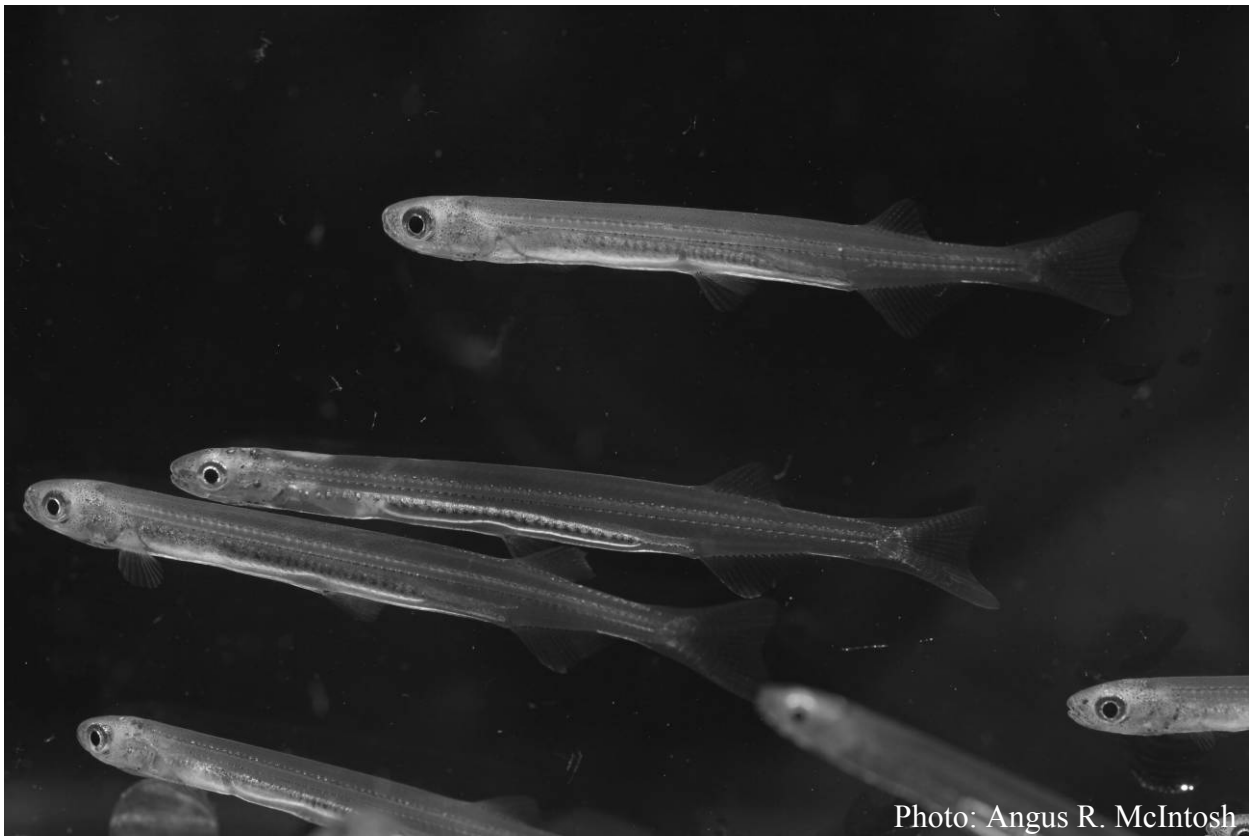
1996). The main issue related with fish passage design is the lack of information about the biology of target fish species. My findings add to the knowledge of the swimming behaviour and climbing capability of a weak-swimming species. Furthermore, the information gathered can be applied in the field: ramp dimensions (i.e., width, length, angle) will need to be adjusted to the flow regime and to the dimensions of the barrier present to facilitate juvenile inanga passage. Incorporating these adjustments will enable upstream passage during low and high flow conditions. To increase the chance of fish passage in large waterways, ramps should also preferably be present along both banks to enable fish to conduct their upstream climb at both sides of the channel (Clay 1995). Fish ramps are frequently installed at considerable distances from, for instance, weirs or dams to keep the ramp angle as shallow as possible. When approaching such a barrier, however, fish often try to find passage directly at the barrier, making it difficult to locate the ramp entrance (Neil Deans, Nelson Marlborough Fish & Game, personal communication). Hence, fish ramps should be installed at the immediate downstream side of a barrier as long as shallow ramp angles can be maintained.

Managers need to carefully adjust the trade-off between ramp length, angle and life stage of the target fish species before installing fish ramps in the field. For weak swimming species, ramps should, if possible, have a maximum angle of 15 degrees over a three metre length, with rough surface materials to reduce water velocities. Although the installation of pools did not significantly improve upstream passage in this study, further research could indicate they may improve passage at ramps longer than three metres. Future experiments could also incorporate juveniles of other migratory species, thereby providing valuable information for the development of optimal passage solutions for target species (MacDonald & Davies 2007). Finally, passage designs should be orientated toward more than just one target species to provide passage for a wide variety of fish species. However, within the group of target species, the weakest swimmer should set the passage requirements (Boubée *et al.* 1999). I studied inanga because it is considered one of the weakest swimming species in New Zealand (Mitchell 1989; Baker 2003;

Nikora *et al.* 2003; Plew *et al.* 2007). Thus, my findings can, and should be, used as base data for passage design, improving upstream passage of many fish species. This might eventually allow upstream fish diversity to expand and fish populations to be sustained in urban waterways.

Chapter Three

JUVENILE GALAXIID SWIMMING BEHAVIOUR AND IMPROVED CULVERT DESIGN TO ASSIST FISH PASSAGE



3.1. INTRODUCTION

Worldwide, stream fish distributions are influenced by urbanisation, including storm water run-off, habitat destruction, increased sedimentation and physical migration barriers (Hogg & Norris 1991; Paul & Meyer 2001; Bernhardt & Palmer 2007). The latter play an especially important role in determining fish distributions as culverts in particular, obstruct fish passage, influencing up- and downstream fish communities (Chapter One; MacDonald & Davies 2007). In the USA culverts create more barriers to fish passage than other structures, but due to cost constraints, corrugated metal pipe culverts are frequently installed instead of more environmentally benign bridges (Johnson & Brown 2000; Gibson *et al.* 2005a). Culverts allow water to flow under roads at stream crossings but they often form a rigid boundary in a dynamic stream environment. As a consequence, they are capable of altering local physical characteristics, hydraulic conditions and biotic linkages within streams. Although engineering guidelines for culvert design and construction are well established, the primary engineering design objective is typically to maximise the hydraulic capacity of the structure for minimum cost, with little attention given to fish passage requirements (Boubée *et al.* 1999).

Fish passage problems at culverts are primarily related to two issues: perching at the culvert outlet (caused by excessive scour below the downstream end of the pipe) and uniformly high velocities within and downstream of the culvert (O'Hanley & Tomberlin 2005; Wargo & Weisman 2006; Kemp & Williams 2008). With regard to the latter, fish prefer to use non-uniform flows as found in un-modified streams during their upstream movements as they provide resting areas of low velocity, enabling fish to move upstream without exceeding their fatigue limit (Bender *et al.* 1992; Peake *et al.* 1997a; Webb 1998; Liao *et al.* 2003a). However, the uniformly high water velocities that fish often encounter at culverts, often exceed their swimming capability, resulting in unsuccessful passage (Hammer 1995). Increased water velocities at the culvert outlet can also cause culverts to perch, thereby increasing the drop-off

height (distance from the water surface to the culvert outlet lip). This commonly results in undercut culverts, presenting fish passage barriers.

Fish trapped below culverts can suffer heavy predation, which is likely to result in disjunct distributions (Stuart & Mallen-Cooper 1999; Coutant & Whitney 2000). Barriers also cause declines of adult stocks and reduce abundance up- and downstream of structures (Joy *et al.* 2000; Baker 2003). In Australia, impacts of dams and weirs on fish migration have been identified as major causes of native fish population declines (Dodd *et al.* 2004) and in the Inner Bay of Fundy, Canada, salmon runs have fallen from 40,000 adults to only several hundred (over a 99% decline) over the past 20 years due to migration barriers (Faragher & Harris 1994).

Many fish species encounter either natural or artificial obstacles during their migrations, resulting in different species developing different methods to negotiate impediments. Some pass barriers by sinusoidal motion (eel species; Anguillidae) or by climbing along wetted margins (juvenile kokopu and koaro, *Galaxias* spp.), some by sucking (lampreys; *Geotria australis)*, jumping (Salmonidae) or burst swimming (inanga adults, *Galaxias maculatus*) (Mitchell 1989; O'Hanley & Tomberlin 2005). Still, despite these morphological and behavioural adaptations, many species are unable to get past in-stream barriers, especially if they are perched, lack wetted margins for fish to rest, or create sustained high velocity flows (Boubée *et al.* 1999; Haro *et al.* 2004). Because of the lack of detailed information on swimming and climbing performance, engineers and managers are frequently unable to assess whether a species of concern will be able to pass such simple structures as culverts (Boubée *et al.* 1999). To be able to remediate passage obstructions, it is important to improve our knowledge about swimming performance and fish behaviour at barriers.

Successful migrations are a vital part of the life history of many fish species, and are undertaken for purposes such as locating suitable habitat, growth, reproduction and food acquisition (Hammer 1995; Nikora *et al.* 2003; House *et al.* 2005). Unimpaired migrations are particularly crucial in the lifecycles of New Zealand fish species as 45 % are diadromous (sea-

migratory) (McDowall 2001). Connectivity between water bodies is not just important for diadromous fish species (e.g., galaxiids, salmonids), but also for potamodromous (migration within freshwaters) fish species such as cyprinids (McDowall 1993, 1998; Knaepkens *et al.* 2004; Knaepkens *et al.* 2006). Thus, successful upstream passage is necessary for maintaining sustainable populations of a wide variety of fish species, highlighting the need for a better understanding of fish behaviour at barriers.

Migratory galaxiids, in particular, represent a very important component of New Zealand's freshwater fish fauna, because of their contribution to the whitebait fishery (McDowall 2000; MacDonald & Davies 2007; James & Joy 2008). However, there is little quantitative information on the heights, slopes and velocities these fish can traverse. Providing upstream access for inanga (*Galaxias maculatus*, Jenyns, 1848) in particular, should be a priority as it is not only the smallest and weakest swimmer, but also the most abundant of the five whitebait species, comprising up to 80 % of the New Zealand (and South American) commercial and recreational whitebait fishery (MacDonald & Davies 2007; Mardones *et al.* 2008).

The inanga is a native New Zealand fish with widespread distribution at latitudes greater than 30° South. It has a circum-Antarctic distribution in both freshwater and coastal waters, and can be found in South America (Chile and Argentina), South Africa, Australia and Tasmania (McDowall 2001). *G. maculatus* is diadromous, meaning its larvae migrate to sea where they spend the winter developing into juveniles (whitebait). During spring each year, whitebait return to fresh water in search of habitat for growth to adulthood (McDowall 2001). As inanga inhabit the lower reaches of most New Zealand rivers, and because many New Zealand cities are located along the coast, their spring migrations frequently encounter culvert barriers (Baker & Hicks 2003). Fish pass issues are therefore a significant impact of urban development in New Zealand (Chapter One) and their effects could be mitigated by retrofitting or designing culverts with fish ramps (Chapter Two; McDowall 2000; Rowe *et al.* 2002).

Fish ramps are hydraulic structures that enable fish passage upstream of an obstruction (Baker 2003; Suren & Elliott 2004). Their efficiency depends on attracting fish that are seeking passage, as well as the safe and speedy transport of fish (Bender *et al.* 1992). However, culvert design does not always take into account the ecology of particular fish species (Bender *et al.* 1992). Most efforts to improve fish passage through culverts have targeted upstream movements of commercially important fish species (e.g., adult salmonids; Stuart & Mallen-Cooper 1999; Kemp & Williams 2008), with the requirements of non-salmonid species and other life stages (e.g., juveniles) being largely un-known (Peake *et al.* 1997a; Knaepkens *et al.* 2006; Richmond *et al.* 2007). Juveniles, in particular, have different requirements for water quality (Coutant & Whitney 2000), depth and/or velocity (Chapter Two; Rowe & Dean 1998; Baker & Hicks 2003) than adults, and may be unable to utilise current culvert designs (House *et al.* 2005). Finally, a lack of understanding of fish behaviour in response to conditions encountered at culverts and ramps often limits the effectiveness of these structures. Moreover, design efforts usually rely on information available about swimming capability from laboratory studies (Kemp & Williams 2008). Although laboratory studies can provide close approximations to conditions fish encounter in nature, there have been no field studies conducted on the swimming performance of juvenile inanga (Chapter Two; Haro *et al.* 2004). My study addressed these issues, focussing on the swimming behaviour of juvenile galaxiids to identify aspects of culvert engineering that will improve fish passage. The main objectives were to investigate: (1) how different culvert designs (i.e., box culverts versus pipe culverts) and (2) habitat variables (e.g., water velocity, riverbed gradient) affect the passage success of juvenile inanga (*Galaxias maculatus*). I also investigated how passage of juvenile inanga at culverts could be improved by *in situ* ramp installation.

3.2. METHODS

3.2.1. Experimental design

I conducted two *in situ* experiments at 13 small (mean diameter < 1.5 m) culverts, within the Nelson City Council (NCC) area, New Zealand (Table 1, Fig. 1). Small culverts are generally difficult to access with heavy machinery to retrofit (e.g., modification of culvert angle) and maintain and therefore require special consideration (e.g., Haro *et al.* 2004). They are also commonly installed in first order streams which are often the only habitat within catchments that can still support the original native fish fauna (Stevenson *et al.* 2008).

In Experiment One I conducted trials at all 13 culverts. I released 50 juvenile inanga (*Galaxias maculatus* Jenyns, 1848) downstream of each culvert to monitor both their swimming behaviour at the culverts and how upstream movement differed with culvert design. In Experiment Two I released 50 juvenile inanga downstream of a ramp installed at five of the 13 culverts that could not be climbed in Experiment One and observed their climbing behaviour before and during the upstream climb.

3.2.2. Study sites

Table 1. The 13 study sites and experimental culverts and their locations within the Nelson City Council area.

Culvert Name	Experiment (I/II)	Waterway	Location (E=East, S=South)
CB	I	Unnamed	E 173°24’41”, S 41°09’45”
D1	I & II	Dodson Valley Stream	E 173°19’31”, S 41°14’36”
J1	I	Jenkins Stream	E 173°15’35”, S 41°18’45”
O3	I	Orphanage Stream	E 173°14’19”, S 41°19’50”
O4	I	Orphanage Stream	E 173°14’27”, S 41°20’80”
P1	I	Poormans Valley Stream tributary	E 173°15’70”, S 41°19’40”
PIT 2	I	Pitchards Stream tributary	E 173°24’54”, S 41°12’11”
T1	I	Todd Valley Stream	E 173°20’28”, S 41°13’50”
T2	I	Todd Valley Stream	E 173°20’29”, S 41°13’26”
W2	I & II	Wakapuaka River tributary	E 173°25’80”, S 41°11’46”
W4	I & II	Wakapuaka River tributary	E 173°25’25”, S 41°11’30”
W5	I & II	Wakapuaka River tributary	E 173°24’60”, S 41°12’52”
W6	I & II	Wakapuaka River tributary	E 173°24’62”, S 41°12’49”

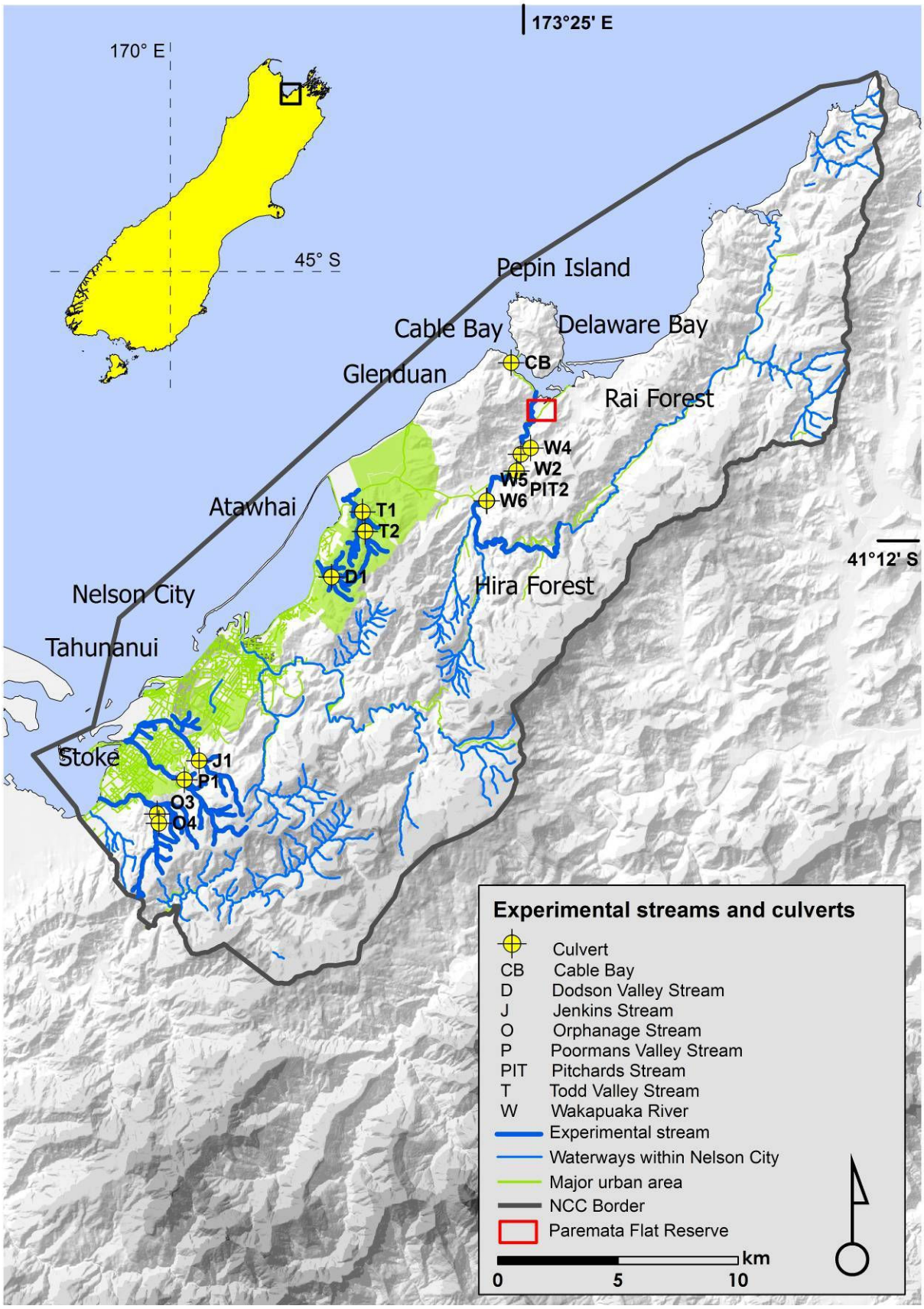


Figure 1. Location of 13 experimental culverts in eight urban waterways within the Nelson City Council area.

3.2.3. Experiment One

3.2.3.1. Experimental design

The 13 culverts used in Experiment One were located on eight urban streams within the NCC area (Fig. 1, Table 1). The ability of 50 juvenile inanga to climb onto each of the 13 culverts was observed for one four hour period between 0800 and 1800 hours daily. The trials were conducted between September and December 2008. For each trial, I installed two nets (2 mm-mesh) to constrain the fish to a section of stream within 1 – 5 m of each end of the culvert. The nets were held in place with steel waratah standards and rocks to prevent fish from escaping (Photograph 1 A) and were regularly cleared of debris. I generally observed fish from downstream of the bottom net to minimise my influence on their swimming behaviour. Fish were counted as they surmounted the culvert outlet and were considered to have successfully passed that point once they were on the culvert outlet lip.

3.2.3.2. Culvert assessment

Culverts were measured and assessed prior to the experiment (Table 2). Drop-off height (the distance from the culvert outlet lip to the water surface) was measured from the outlet lip of the culvert to the water surface below. I measured water velocity (Marsh-McBirney, Flo-Mate 2000 meter) one metre below and above each culvert (three times over the stream width) at 0.4 depth. Barrel velocities (water velocity within the culvert) were measured 0.3 m inside each end of a culvert at 0.4 depth. Stream width (three measurements over a one metre stream reach) and depth (three measurements across the stream) were recorded with a tape measure one metre above and below each culvert. Riverbed gradients and culvert angles were measured with an inclinometer.

3.2.3.3. Fish capture and maintenance

Migrating inanga juveniles (fork length <60 mm) were caught for the experiment from a local stream at the Paremata Flats Reserve in the Wakapuaka River catchment (Fig. 1) with a whitebait set net (2 mm-mesh) and transported to the culvert streams. At the culverts, I kept fish in the stream over night (a minimum of twelve hours) in live boxes (Photograph 1B) to acclimatise before the experiment was commenced the next day. At the completion of each experiment the fish were released in the experimental stream upstream of the culvert.

Table 2. Characteristics of 13 culverts assessed and used for the experiments within the Nelson City Council area with their mean values (±S.E.) and the number of juvenile inanga (out of 50) that passed the entrance of the culvert (outlet) during Experiment One. Perched = culverts have an excessive scour at the outlet; C=culvert, d/s = downstream, u/s = upstream, vel=velocity.

Culvert	Perched (Yes/No)	Design (Pipe/Box)	Barrel vel (m/s)	Vel outlet (m/s)	Vel inlet (m/s)	Depth in C (m)	Depth d/s C (m)	Depth u/s C (m)	Stream width d/s C (m)	Stream width u/s C (m)	C width (m)	C height (m)	C length (m)	Drop-off (cm)	Riverbed gradient d/s (°)	Passed outlet (Fish no.)
CB	N	Box	0.1	0.3	0.2	0.03	0.10	0.09	1.5	1.5	2.00	0.85	10.00	10	1	41
D1	Y	Single Pipe	0.5	0.5	0.5	0.05	0.05	0.06	1.4	1.2	1.00	1.00	10.20	95	7	0
J1	N	Trough Box	0.5	0.5	0.6	0.04	0.06	0.10	3.0	2.5	4.00	1.50	13.50	3	3	35
O3	N	Single Pipe	0.4	0.1	0.4	0.03	0.12	0.20	1.0	1.5	1.00	1.00	17.00	26	3	2
O4	Y	Single Pipe	0.2	0.2	0.5	0.03	0.08	0.08	1.5	1.5	0.60	0.60	17.40	40	4	0
P1	Y	Single Pipe	1.1	0.2	0.1	0.04	0.10	0.06	1.5	1.5	0.85	0.85	7.50	5	8	0
PIT2	Y	Single Pipe	0.6	0.8	0.5	0.03	0.08	0.06	4.0	2.5	0.35	0.35	5.80	15	5	0
T1	N	Open Bottom	0.3	0.4	0.4	0.04	0.04	0.07	2.8	1.0	2.10	2.00	9.20	10	3	45
T2	N	Single Pipe	0.5	0.3	0.04	0.04	0.12	0.10	1.0	3.5	1.45	1.45	9.85	0	3	40
W2	Y	Single Pipe	0.5	0.1	0.2	0.03	0.05	0.08	0.8	0.9	0.95	0.95	14.50	30	6	0
W4	Y	Stacked Pipe	0.6	1.0	0.1	0.06	0.10	0.07	0.6	2.6	0.37	0.37	6.85	10	4	0
W5	Y	Single Pipe	0.5	0.4	0.4	0.04	0.08	0.15	0.5	0.5	0.75	0.75	4.60	15	5	0
W6	Y	Single Pipe	0.5	0.5	0.4	0.04	0.05	0.10	0.5	0.5	0.75	0.75	5.40	13	5	0
Mean			0.49	0.39	0.32	0.04	0.08	0.09	1.54	1.62	1.24	0.95	10.13	20.90	4.30	14.38
(S.E.)			(0.06)	(0.07)	(0.04)	(0.002)	(0.01)	(0.01)	(0.30)	(0.25)	(0.27)	(0.12)	(1.24)	(6.90)	(0.54)	(5.38)

3.2.3.4. Statistical analyses

I used paired *t*-tests and analyses of variance (ANOVA) to evaluate differences in habitat characteristics between eight perched and five non-perched culverts and to determine whether there were differences between physical characteristics of culverts and the number of fish that passed. All statistical analyses were conducted with STATISTICA 8.0 unless otherwise stated.

3.2.4. Experiment Two

3.2.4.1. Experimental design

To assess whether fish passage could be improved, I installed fish ramps (Photograph 1B) at five of the 13 culverts where fish were unable to surmount the culvert outlet in Experiment One (Table 1, Fig. 2). The main factor influencing passage success was expected to be ramp angle, which is a product of ramp length and height of the barrier. Apart from installation of the ramp at five culverts, the duration of the experiment, fish capture, fish maintenance during and after the experiment, and culvert assessment were as for Experiment One. All fish were used once only. The ramp as built in my experiments (Chapter Two, Chapter Three) has not been used in any previous studies, however, the basic aspects of its design were from previous work (e.g.; Baker & Boubée 2006).

Photograph 1. Experiment One involved the release of 50 juvenile inanga downstream of a culvert without a ramp installed to observe fish swimming behaviour and climbing ability onto a culvert (A) (Culvert T1). At culverts where fish were unable to pass in Experiment One, ramps (arrow) were installed (B) in Experiment Two to determine whether upstream passage improved with ramp installation (Culvert W4). Nets installed up- and downstream of each culvert stopped fish escaping from the experimental reach (A,B). Fish were kept in live boxes until the start of the experiment (B).

3.2.4.2. Ramp hydraulics

The three-metre experimental ramp was 300 mm wide and v-shaped in cross-section (Fig. 2). As fish prefer rough substrata for their upstream climb (Baker & Boubée 2006), I used Astroturf on the surface to offer fish suitable conditions to rest during their climb (as in Chapter Two). I did not control water velocities during the experiments, thus, the conditions fish encountered were realistic. Ramp angles were determined by stream bed gradient and the drop-off height at the culvert outlet, but were no steeper than 20° at any culvert, an angle considered to be traversable by *G. maculatus* (Chapter Two; Baker & Boubée 2006).

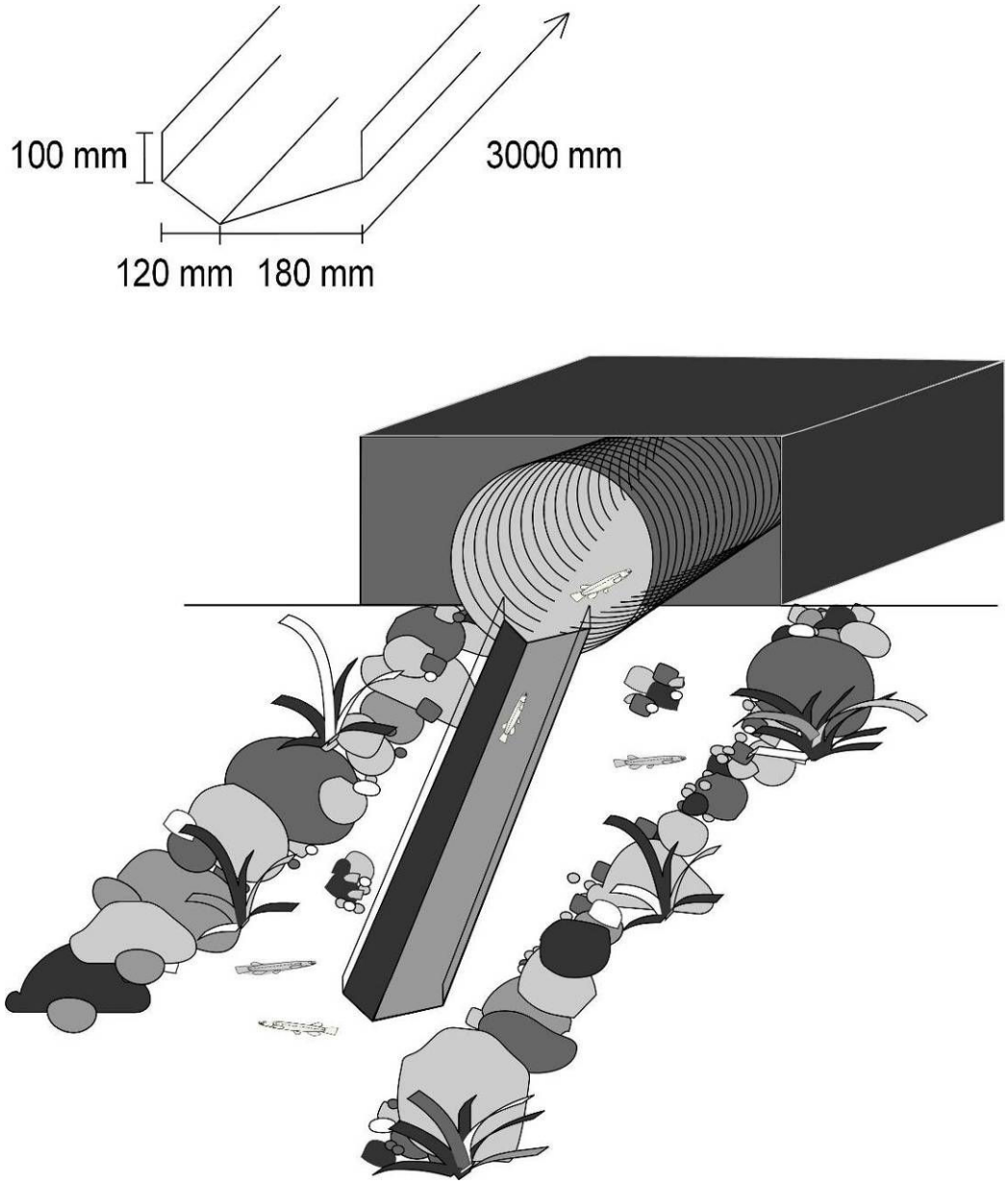


Figure 2. Physical dimensions of the ramp and its positioning with respect to the culvert outlet in the field.

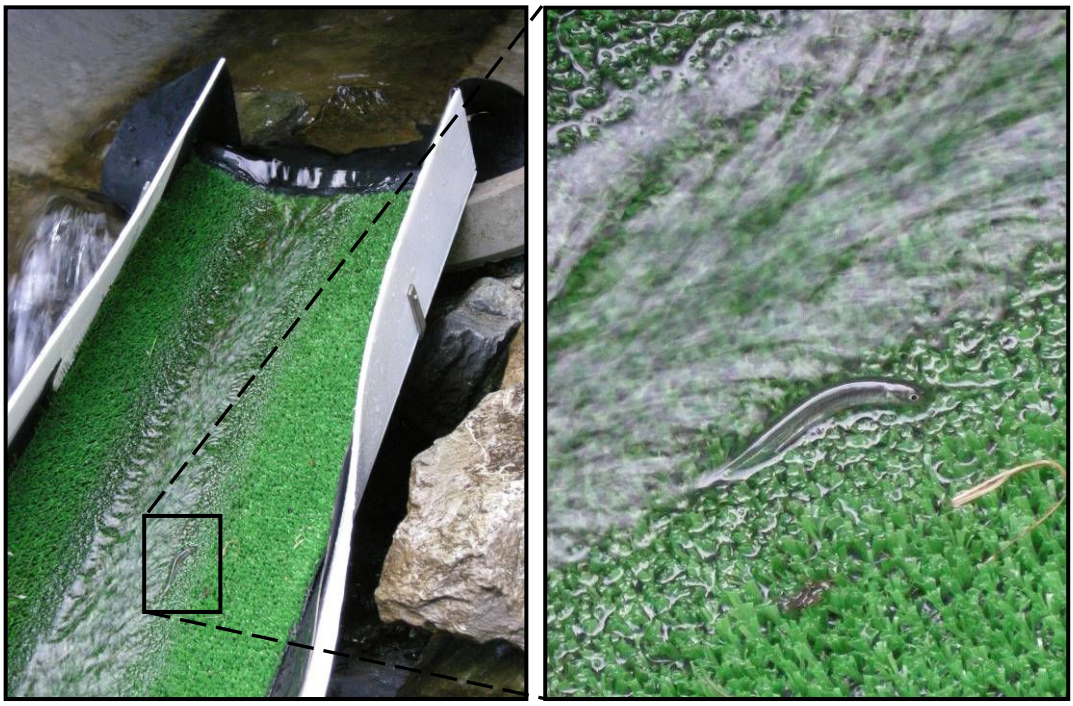
3.2.4.3. Statistical analyses

I conducted two linear regressions, one to evaluate the relationship between ramp angle (untransformed) and the number of fish that passed (log-transformed), and the other to investigate the relationship between ramp angle and ramp water velocities. I also conducted paired *t*-tests to evaluate whether habitat variables recorded on the culverts differed between Experiment One and Two.

3.3. RESULTS

3.3.1. Fish behaviour

It took fish approximately 30 min to acclimatise to their new environments within the experimental stream reach in both experiments. When first released into the water, fish tried to hide and seek cover, mostly as a shoal. After 30 min the group split up and fish spent more time at the same spot and started feeding. After approximately one hour, fish began to attempt the upstream climb in both experiments. Before and during their ascent, fish swam back and forth, possibly until they found velocities within their swimming ability and subsequently attempted to climb for the remaining time. In Experiment One, fish burst swam up the culvert outlet, often at places where substrate or algae were attached to the surface. Fish attempted to climb the culvert in groups rather than alone (minimum of two individuals). Although observations were sometimes difficult at the immediate culvert outlet, I was able to count fish as soon as they were onto the culvert lip. In Experiment Two, most of the fish tried to get up the ramp by burst swimming before they rested on the wetted margin provided by the v-shaped ramp (Photograph 2). Fish rested several times before they were either washed back downstream or continued their upstream climb.



Photograph 2. A juvenile inanga resting on the wetted margin of the ramp installed at a culvert in the field (Culvert D1).

3.3.2. Experiment One

Culvert design affected water velocity, and the habitat within the structure. Pipe culverts generally offered less habitat than larger open-bottom box culverts due to their smaller diameters (Photographs 3A & B).

To determine whether fish passage success depended on habitat variables (i.e., water depth and velocity, stream width) I tested for differences in conditions upstream and downstream of the culverts (Table 3).

Photograph 3. Pipe culverts (A) confine stream width, frequently resulting in uniform and increased water flow and lower availability of in-stream habitat (Culvert P1). Open-bottom box culverts (B), in contrast, usually retain the original riverbed width, and have variable flow conditions offering more diverse habitat (e.g., substrate) within the structure (Culvert T1).

Table 3. Mean values (±S.E.) of three habitat variables measured 1 m upstream (u/s) and downstream (d/s) at the 13 culverts used in Experiment One. Paired *t*-tests compare u/s and d/s values; df= degrees of Freedom, P=Probability.

	Unit	**Mean u/s (± S.E.)**	**Mean d/s (± S.E.)**	***t*- value**	**df**	***P*- value**
Water depth	m	0.09 (0.01)	0.08 (0.01)	-1.36	12	0.19
Stream width	m	1.62 (0.25)	1.54 (0.30)	-0.26	12	0.79
Water velocity	m/s	0.32 (0.05)	0.39 (0.08)	0.92	12	0.37

As physical characteristics upstream and downstream of the 13 culverts did not vary significantly (Table 3), I tested for other variables that could have influenced fish passage at the culverts. Perching was critical, as no fish successfully negotiated any of the perched culverts used in Experiment One (Table 2). The 95% confidence intervals for the percentage of fish that passed non-perched culverts (44 – 84 %) indicated a significant difference between perched and non-perched treatments (Fig. 3).

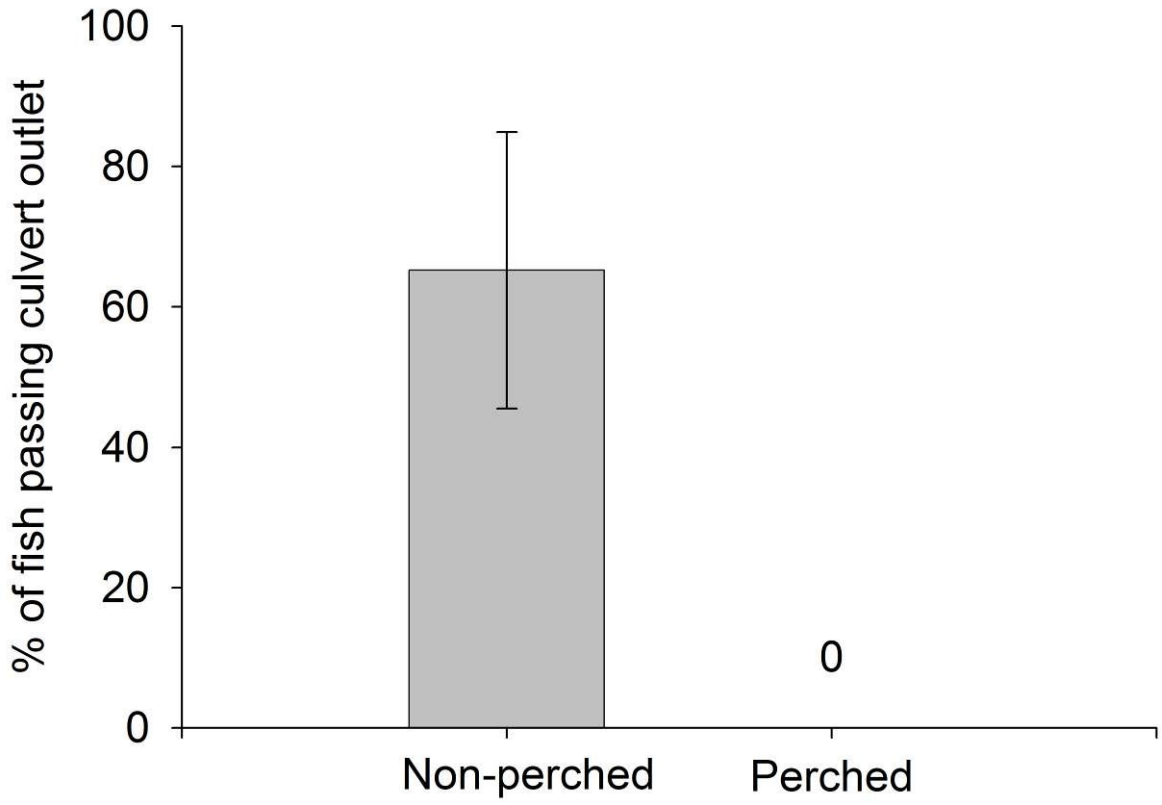


Figure 3. The percentage of fish that passed perched (*n* = 8) and non-perched (*n* = 5) culverts. The error bar indicate 95% confidence interval.

To test whether passing a culvert depended solely on perching, or whether other factors could have been involved, I compared habitat variables and physical dimensions of five non-perched and eight perched culverts (Table 4). Barrel velocity, riverbed gradient, culvert width and height, all differed significantly between perched and non-perched culverts. Therefore, I concluded, that perching was the main reason why fish did not get up a culvert, but that high gradient and high barrel velocity may also have played a role (Table 4).

Table 4. Mean (±S.E.) habitat and physical dimensions measured at eight perched (undercut) and five non-perched culverts (C). Paired *t*-tests compare perched and non-perched values. Significant values ($P \leq 0.05$) are highlighted in bold, df = 4.

	Perched		Non-Perched		*t*-value	*P*-values
	$\bar{x}$	S.E.	$\bar{x}$	S.E.		
Water velocity outlet (m/s)	0.45	(0.11)	0.30	(0.05)	1.09	0.33
Water velocity inlet (m/s)	0.32	(0.06)	0.31	(0.08)	0.02	0.97
Barrel velocity (m/s)	0.90	(0.14)	0.24	(0.04)	2.74	**0.05**
Water depth in C (m)	0.04	(0.003)	0.03	(0.002)	0.53	0.62
Water depth d/s C (m)	0.07	(0.01)	0.08	(0.01)	0.91	0.41
Water depth u/s C (m)	0.08	(0.01)	0.11	(0.02)	0.64	0.55
Stream width d/s C (m)	1.33	(0.40)	1.86	(0.39)	1.09	0.33
Stream width u/s C (m)	1.38	(0.29)	2.00	(0.40)	0.72	0.50
Riverbed gradient (°)	5.50	(0.50)	2.60	(0.36)	4.70	**0.01**
C width (m)	0.70	(0.08)	2.11	(0.51)	3.33	**0.02**
C height (m)	0.70	(0.08)	1.37	(0.19)	2.91	**0.04**
C length (m)	9.03	(1.65)	11.91	(1.47)	2.50	0.06
Drop-off outlet (cm)	27.80	(10.39)	9.80	(4.00)	0.97	0.38

3.3.3. Experiment Two

To test whether the impact of perching could be mitigated, I installed a fish ramp to facilitate passage at five of the eight perched culverts. Because I conducted Experiments One and Two one month apart (although maximum fish size (fork length < 60 mm) stayed the same), I tested whether significant differences between physical habitat variables measured at culverts during the two experiments could have affected the results of Experiment Two. Physical variables during Experiment Two were similar to those experienced at the same sites in Experiment One (Table 5).

Table 5. Paired *t*-tests comparing physical variables measured at the time of Experiments One and Two at five culverts (C); u/s = upstream, d/s = downstream, df = 4.

	***t* - value**	***P* - value**
Water velocity in C (m/s)	0.155	0.88
Water velocity at outlet (m/s)	1.426	0.23
Water velocity at inlet (m/s)	0.151	0.89
Water depth d/s C (m)	1.121	0.33
Water depth u/s C (m)	0.666	0.54
Stream width d/s (m)	0.001	1.00
Stream width u/s (m)	0.001	1.00
Drop-off height (m)	0.001	1.00
Riverbed slope (°)	0.408	0.70

No fish passed any of the five culverts selected for Experiment Two during Experiment One. With the ramp installed, upstream passage was 44 % with the 95% confidence interval for the number of fish passing not including zero (20.91 – 66.28 %). Hence, the ramp significantly improved fish passage at perched culverts.

I also found that water velocity increased with ramp angle (linear regression: $F_{1,4} = 14$, $P<0.05$, $R^2=0.82$) and that significantly fewer fish passed the ramp as ramp angle increased (linear regression on log-transformed number of fish: $F_{1,4} = 25.11$, $P<0.05$, $R^2=0.89$; Fig. 4).

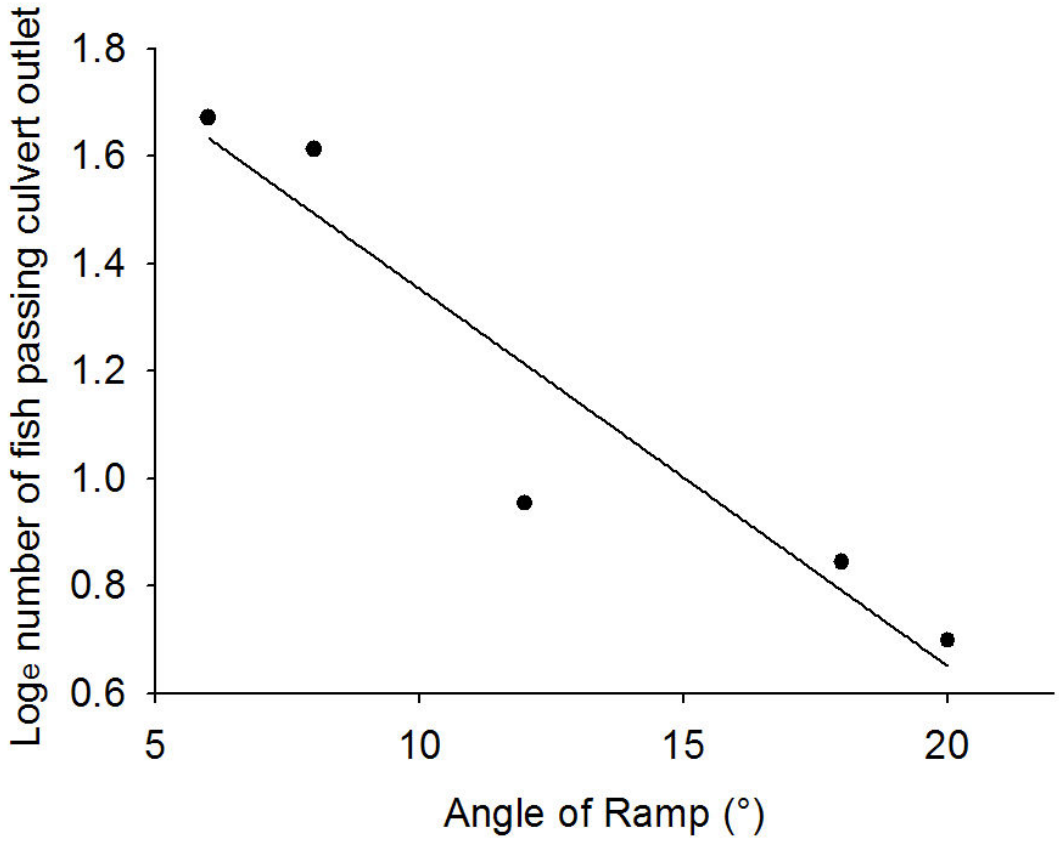


Figure 4. Relationship between the number of fish passing 5 culvert outlets and the angle of the ramp (°) (Log_e number of fish passed = - 0.07e $^{ramp\ angle}$ + 2.05).

3.4. DISCUSSION

To maintain freshwater fish populations in urban stream environments, biological processes (feeding, exploration or social interactions) and physical processes (turbulence and migration barriers) need to be considered (Beamish 1970; Webb & Gerstner 2000; Cea *et al.* 2007). Physical processes play a particularly important role, as poorly designed, installed and/or maintained migration barriers such as culverts can severely delay or block upstream movement of fish (Chapter One).

My results indicate that one of the main issues influencing fish passage at 13 Nelson sites was perching due to inappropriate culvert installation. None of the 400 fish in my study were able to pass any of the eight perched culverts. Perched culverts were narrow and small, and occurred on streams with steep bed gradients, potentially explaining why the culverts were initially perched. If culvert gradient is shallower than riverbed gradient, the riverbed is likely to scour below the culvert outlet (Stevenson *et al.* 2008). Barrel velocities were also significantly higher in perched than non-perched culverts and results, which happens if culvert width is narrower than that of the natural stream bed. Water velocity plays a crucial role in determining fish passage as indicated in Chapter Two, where I showed that increased water velocity restricted the upstream passage of juvenile inanga. However, once the perching effect was removed by ramp installation, fish were able to pass the culverts in Experiment Two, demonstrating that the effects of perching can be overcome by retrofitting ramps.

Only one laboratory study has investigated the climbing behaviour of juvenile inanga up ramps (Baker & Boubée 2006) and my study is the first to conduct *in situ* ramp experiments with whitebait at culverts. Successful fish passage at ramps depended on ramp water velocities, with fewer fish passing at high velocities. House *et al.* (2005) found that velocity patterns are generally linked with the swimming ability of small fish/juveniles at culverts. This concurs with my previous laboratory study where fewer juvenile inanga passed

culverts successfully as ramp water velocity increased (Chapter Two). Moreover, water velocities exceeding 0.3 m/s are also likely to restrict passage of other juvenile fish species such as shortfin eel (*Anguilla australis*), banded kokopu (*Galaxias fasciatus*), common smelt (*Retropinna retropinna*) and common bully (*Gobiomorphus cotidianus*) (Mitchell 1989). Although Mitchell (1989) investigated the swimming ability of these fish species in a horizontal level flume, and therefore disregarded the relationship between water velocity and ramp/culvert angle, his results indicate that many species have limited abilities to withstand even modest velocities. A sensible trade-off between ramp and culvert angle and ramp and barrel velocities, is therefore, crucial for successful ramp and culvert design (Chapter Two). Average water velocity restricting upstream fish passage at the ramps in my experiments (including the angle effect) was 0.32 m/s. To guarantee successful ramp negotiation by several weak swimming fish species, water velocity should be kept below 0.3 m/s at ramps and culverts.

Ramp angle, probably the ultimate controller of fish passage, should be kept as shallow as possible to reduce water velocity, thereby enabling fish to pass. The most suitable angle for juvenile inanga to pass, lies between ten and 15 degrees on a three metre long ramp (Chapter Two). No fish successfully negotiated the ramp at angles greater than 15 degrees in the research reported in Chapter Two. In the present study, however, fish were able to pass over a three metre ramp at a 20 degree angle in the field, showing that studies conducted under controlled conditions, such as in the laboratory, do not necessarily mirror conditions in natural environments.

3.4.1. Implications of ramp installations

Many existing culverts are non-functional or in disrepair and could be easily breached to form routes of passage, provided that fish are capable of traversing the resulting velocity barriers (Schilt 2007). The culverts tested in my study were mainly of small diameter (< 1.5 m) and difficult to retrofit (i.e., inaccessible with large machinery, Haro *et al.* 2004). Nevertheless, small culverts are particularly important for fish passage as the first order streams in which they are installed are often the only habitats within catchments that can still support the original fish fauna (Stevenson *et al.* 2008). Councils should prioritise the retrofitting of small to intermediate culverts in their regions, and consider installing culverts, such as box culverts with wide barrel widths. This will not only help overcome increased barrel velocities, but also avoid perching.

If perching is unavoidable or already occurring, fish ramp installation could considerably improve upstream fish passage. The kind of fish ramp design needs to be dictated by the swimming abilities of the target species. Salmonids, for example, have different passage requirements than galaxiids, as they traverse barriers by jumping (Ojanguren & Brana 2003; Naughton *et al.* 2007; Kemp & Williams 2008), whereas galaxiids either burst swim (inanga) or climb (koaro) past barriers (Haro *et al.* 2004).

The angle of the ramp also directly influences fish passage success, as it determines water velocity on the ramp. Angles should be as shallow as possible, but certainly no steeper than 20 degrees for a three metre long ramp to enable fish to pass successfully. Ramps longer than three metres may have to be set even shallower than 15 degrees and future studies could incorporate this assumption in their design. Local authorities could use my findings at comparable culverts when installing fish ramps to enable the passage of similar fish species (i.e., juvenile bully species; Mitchell 1989). Measures of success could be used to improve knowledge about fish passage design at barriers.

International studies have shown that rock-ramps are suitable to overcome physical migration barriers as they offer fish near natural stream characteristics within the structure itself, resulting in successful fish passage (Beatty *et al.* 2007). A "hybrid"-design between my ramp and rock ramps could provide passage past small barriers for New Zealand fish species, improving upstream abundance of barriers.

Finally, urban areas lie at the intersection of human and ecological systems (Paul & Meyer 2001). Hence, culverts and other road crossings installed as part of urbanisation will increasingly modify waterways, worldwide (Haro *et al.* 2004; Blakely & Harding 2005; Blakely *et al.* 2006). It is therefore important to increase the scope of studies measuring fish passage (O'Hanley & Tomberlin 2005; Baker & Boubée 2006). If we are to succeed in that often-stated goal by ecologists and engineers of, "incorporating humans as components of ecosystems", urban streams can no longer be ignored (Paul & Meyer 2001; Schilt 2007). Amongst other factors important for fish diversity in urban streams, such as water quality, retaining fish passage in urban areas through "smart" culvert design will not only offer ecologists the opportunity to maintain fish diversity in urban waterways and to test concepts of urban stream restoration, but will also enable people to experience an easily accessible piece of nature within cities.

Chapter Four

RECOMMENDATIONS AND CONCLUSIONS

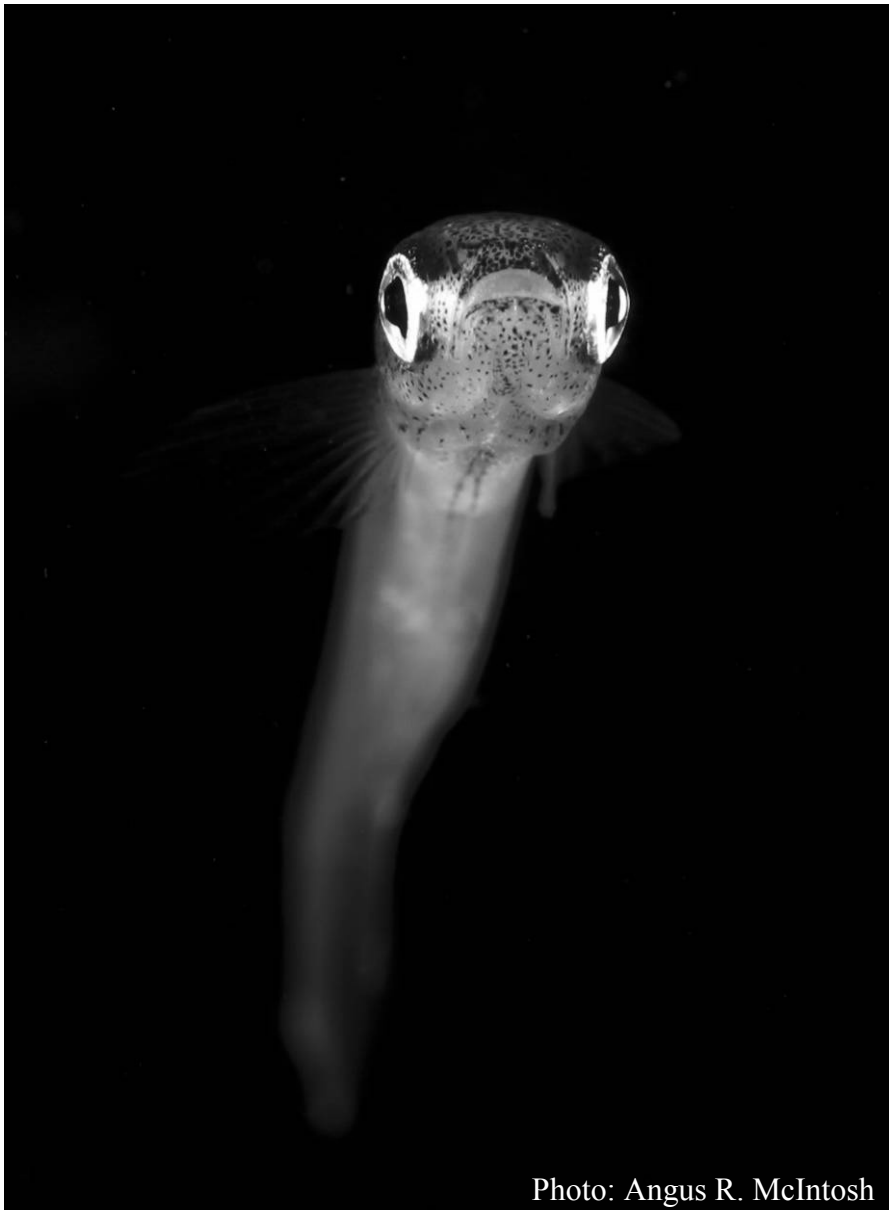


Photo: Angus R. McIntosh

4.1. INTRODUCTION

In lowland rivers many non-salmonid fish species such as cyprinids (e.g., barbel, *Barbus barbus*) or galaxiids (e.g., inanga, *Galaxias maculatus*) exhibit substantial seasonal migrations to find appropriate spawning habitat (Baras & Cherry 1990; Lucas *et al.* 1999; McDowall 2007). These migrations, as well as fish species distribution and community composition, are controlled by a series of interacting biotic (i.e., foraging, predation, competition) and abiotic (i.e., water velocity) factors within a geographic region (McIntosh & McDowall 2004; Pavlov *et al.* 2008). It is the task of freshwater ecologists to unravel this spatial and temporal complexity to determine the drivers that influence fish migrations and distributions. Many of these drivers are still unknown, but the effect of urbanisation is likely to be one of the most important. My study has revealed details of the effects of urbanisation on fish communities and added to knowledge of how some of these effects can be overcome. My data can be transferred to other parts of New Zealand and other countries in the Southern Hemisphere (Australia, Chile) where inanga is found, and also to other fish species with similar swimming capabilities (e.g., bully species).

This chapter summarises the key issues influencing fish distribution within urban waterways that were identified in the previous chapters, and provides recommendations for managers and local authorities on how to prevent and remediate some of these issues. I will also point out potential future research needs and highlight the applicability of my findings for urban fish community restoration.

Urbanisation, in general, can have major impacts on fish distribution and community composition (Chapter One). For example, fish species, such as giant kokopu or dwarf galaxias that are susceptible to poor habitat quality (Bonnett & Sykes 2002; Baker & Smith 2007) were rare or absent within the Nelson City Council (NCC) area, whereas more tolerant fish species such as eels (Hicks & McCaughan 1997; Glova *et al.* 1998) were abundant. Moreover, communities that contained sensitive fish species were associated with native forest cover, whereas tolerant fish

species were associated with urban and agricultural land-use types. There were clear-cut groups of fish communities according to land-use type for the two geographically different regions I analysed (the Nelson City Council and the upper South Island), and for analyses conducted at the reach scale and the catchment-scale. Urbanisation affected fish distribution and communities at both, small-scale and large-scales and was an important driver influencing fish distribution throughout the upper South Island. Perched (undercut) culverts in particular influenced fish species distribution and abundance in the Nelson City Council area (Chapter Three). Inland penetration by inanga, for instance, was significantly reduced by culverts in urban Nelson (Chapter One). Previous studies have already highlighted the susceptibility of inanga to migration barriers (e.g., Baker 2003; Baker & Boubée 2006), but my results revealed that even fish species known to be good climbers (i.e., banded kokopu) were, eventually impacted by the accumulation of culverts found in urban environments (Chapter One).

To further investigate this finding, I studied the swimming ability of inanga in the laboratory (Chapter Two) and found that juvenile inanga are weak swimmers, making them vulnerable to the increased water velocities and steep gradients occurring at physical impediments like culverts. To mitigate these impacts, I tested the effect of pools as a resting opportunity on a ramp. Although pools have enhanced fish passage in previous studies (Rajaratnam *et al.* 1997; Kim 2001), juvenile inanga passage success did not significantly increase in my study. My findings also indicated that ramp angle and fish size were important determinants of passage. More fish passed on shallow ramp angles and larger fish were more successful. To test whether my laboratory findings could be applied *in situ*, I installed the ramp at five different culverts (Chapter Three).

The installation in the field considerably improved the number of fish getting onto culverts, potentially enabling them to reach upstream habitats for reproduction and growth (Chapter Three). Similar to the laboratory results in Chapter Two, water velocities and ramp angle determined fish passage up the ramp under field conditions.

My findings looked at the effects of physical barriers as part of urbanisation and summarised potential improvement techniques. The remediation of barriers is particularly important, as decisions over culvert installations or retrofitting have cost and logistical implications. To successfully design road crossings and fish ramps, managers need information on potential remediation techniques and fish behaviour.

Prior to the installation of a culvert or a fish ramp there are two main issues that need to be considered. Firstly, managers need to determine the range of fish species that are potentially affected by the culvert or ramp installation. This is crucial as every fish species behaves differently depending on the type of barriers. To provide passage for most species present in an area, the target species should be the one with the lowest swimming ability, such as inanga and common bully in the Nelson City Council area (Boubée *et al.* 1999; Larnier 2002). By designing culverts that enable these species to pass, fish passage will also be provided to stronger swimmers such as redfin bully (*Gobiomorphus huttoni*) (Baker & Boubée 2006).

Secondly, managers and councils should be aware of the quantity and quality of suitable habitat for the target fish species available upstream and whether it is feasible and necessary to go through the often complex process of culvert retrofitting and fish ramp installations. Whether upstream habitat is suitable for the target fish species can be determined by, for instance, detailed surveying of the upstream habitat (e.g., Doehring & McIntosh 2008) or by predictive modelling (e.g., Joy & Death 2000, 2004). Sometimes however the location of a culvert does not always allow managers to redesign a culvert that meets the requirements of the target fish species, due to the topography of the location, for example. In this case, managers should focus on the installation of fish ramps. Below I firstly outline the problems associated with culverts and potential remedies, before discussing issues related to mitigation techniques associated with ramp installations.

4.2. CONSIDERATIONS FOR CULVERT INSTALLATION

In general, fish ramp installation could be avoided if culverts are designed and installed appropriately in the first place. Which type of culvert to install is an important point, and my results can be used to guide this decision. I indicated in Chapter Three that (open-bottom) box culverts are less likely to influence fish movement, and are 'fish friendly'. They are similar to a bridge in their hydraulic characteristics (Boubée *et al.* 1999), and provide habitat within the structure (Fig. 1; Boubée *et al.* 1999). Moreover, box culverts also frequently have the same width as the stream bed, resulting in the same water velocities as the natural flow. A shift in installation from pipe culverts to box culverts should in most cases improve fish passage considerably and is recommended (Cotterell 1998).

Increased barrel velocities are caused if culvert widths are less than that of the natural stream bed (Cotterell 1998; Johnson & Brown 2000; Wargo & Weisman 2006; Stevenson *et al.* 2008). Barrel velocities were significantly greater in pipe than in box culverts in the NCC area, likely influencing passage success (Chapter Three). Small culvert widths as found in stacked pipe or pipe culverts (Fig. 1) especially at times of high flow, are likely to cause increased water velocities, obstructing upstream fish passage (Stevenson *et al.* 2008). Engineers and fisheries biologists have known for years that velocity patterns are intimately linked with the ability of small fish (juveniles) to get onto and move through culverts (House *et al.* 2005). Thus, they should install culverts with wide stream bed widths, such as open-bottom box, trough box or simple box culverts (Fig. 1).

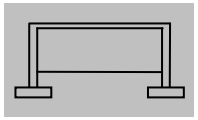
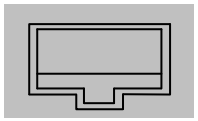
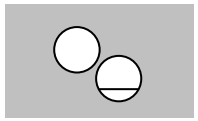
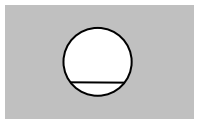
Type of culvert	Risk of fish passage restriction	Hydraulic considerations	Remediation techniques
Open Bottom Box 	• Low risk • In-stream habitat available	• Maintains natural channel width	• Regular maintenance (i.e., removal of debris)
Trough Box 	• Low risk • Suitable for low flow conditions	• Maintains natural channel width • Trough can fill with bed load and create maintenance problems	• Regular maintenance (i.e., removal of debris)
Simple Box 	• Low risk • In-stream habitat available • Limits fish passage during low flow	• Maintains natural channel width	• Installation of a low flow channel at culvert bottom
Stacked Pipe 	• Medium to high risk • Allows fish passage during a wider range of flows than pipe culverts • Limited in-stream habitat	• Constricts natural channel width • Creates high flow velocities • Increased chance of perching	• Installation of wide diameter pipes • Shallow culvert angle • Raising of the stream bed level to culvert height
Single Pipe 	• High risk • Limited in-stream habitat	• Constricts natural channel width • Creates high flow velocities • Increased chance of perching	• Installation of wide diameter pipes • Shallow culvert angle • Raising of the stream bed level to culvert height

Figure 1. Culvert shapes occurring in the Nelson City Council Area and associated considerations for fish passage, hydrology and remediation (modified from Adams & Whyte 1990; Warren 1998; Parker 2000; James & Joy 2008).

Increased velocities also occur when pipe culverts are installed at riverbed gradients steeper than that of the culvert (Fig. 2), and this drives initial culvert perching. In Chapter Three I showed that the riverbed gradients of perched culverts within the NCC area were significantly steeper than of non-perched culverts, potentially influencing fish passage.

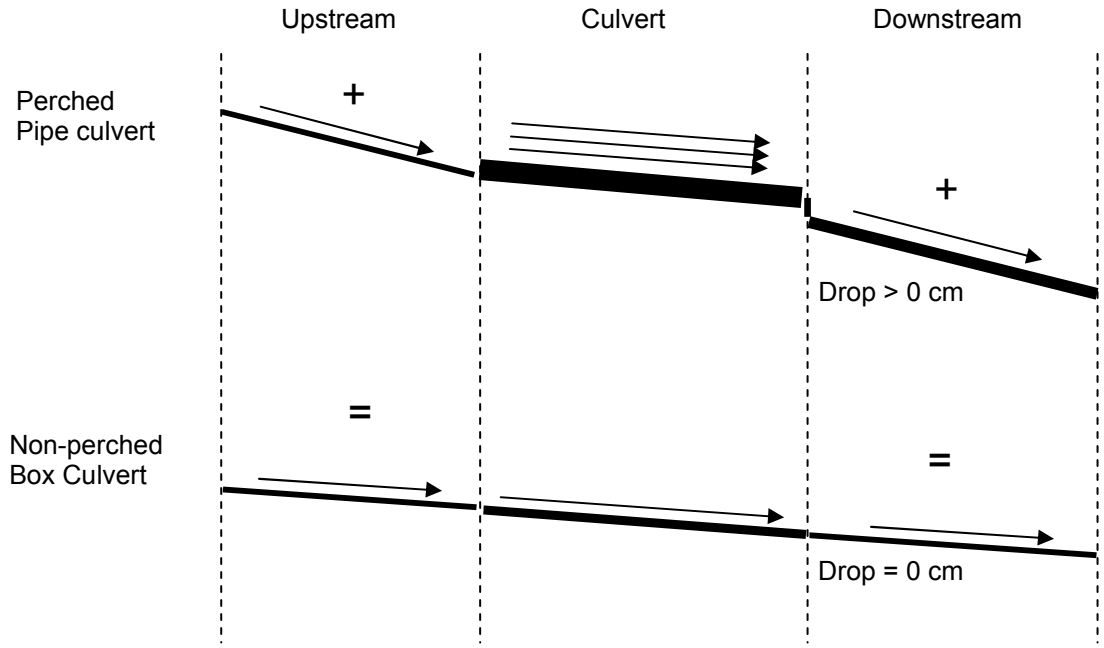


Figure 2. Differences between the riverbed gradients at perched and non-perched culverts and the resulting changes in water flow and velocities (modified from Fitch 1996); + River bed gradient steeper than culvert, = River bed gradient same as culvert, ▬▬ = Depth of flow, ⟶ = Flow velocity.

Perching was one of the main drivers determining fish passage in the NCC area (Chapter Three) and should be avoided. To avoid perching in the first place, culvert gradient and riverbed gradient should be similar (Fig. 2) with culvert gradients recommended to be, in general, not greater than 1 – 2 % (Clay 1995; Fitch 1996; Stevenson *et al.* 2008). Furthermore, perching is also caused by stream channels that are likely to degrade due to the riverbed geology of the construction site (Cotterell

1998). Although I did not investigate the geology of the Nelson City Council area, this factor should be considered before designing and constructing culverts. To account for the different riverbed geology in an area, all culverts should be designed with a specific minimum countersunk dimension (i.e., 20 % of the culvert diameter or at least 15 cm to prevent perching (Fitch 1996; Cotterell 1998). If steep culverts cannot be avoided or cannot be retrofitted to fish-friendly designs, fish ramps should be installed.

4.3. CONSIDERATIONS FOR FISH RAMP INSTALLATIONS

I showed that the ramp installation at perched culverts in Chapter Three improved fish passage considerably as soon as the ramp was installed. However, the lack of past research on the biology of native fish (i.e., the sizes of fish and their life stages that are migratory, fish response to obstacles), has led to the construction of relatively ineffective fish ramps (Stuart & Mallen-Cooper 1999). I therefore want to highlight the most important issues that engineers, ecologists and managers need to consider when designing fish ramps.

In general, attraction to surmount a barrier/ramp is critical and depends on species behaviour and motivation (Bender *et al.* 1992). However, there are factors that can increase the attraction for fish to surmount a ramp (Bunt 2001). Often, it is the physical characteristics at the ramp that increase or decrease attraction. My study indicated that ramp gradient, length and water velocity were the main variables influencing passage success in both, the laboratory as well as in the field, and a realistic trade-off between these three factors is crucial to design successful fish ramps (Fig. 3). In my studies, ramp gradient played an important role as more fish passed the ramp when the angle was shallow (Chapter Two and Tree). Fish passage success can therefore be improved by keeping the fish ramp angles as shallow as possible. My study indicated that the ideal ramp angle should lie between 15° and 20° at a three metre long ramp, but no greater than 20° if weak swimmers are targeted (Chapter Two).

It needs to be considered that even with shallow ramp angles some fish may be unable to pass the ramp if the climbing distance exceeds their burst swimming ability (Peake *et al.* 1997b; Boubée *et al.* 1999). In these situations resting areas for fish during the climb are recommended. Although pools as resting opportunities did not significantly improve juvenile inanga upstream passage in my study, a different type of resting zone might improve upstream passage. For instance, a 50 – 100 mm zone of wetted margin on either side of the ramp creates resting opportunities for New Zealand fish species which use the surface tension to sit on (Boubée *et al.* 1999). The availability of a wetted margin as a resting opportunity is especially important when water velocities exceed the species' swimming ability.

I showed in Chapter Three that water velocities are a function of the ramp slope (Fig. 3) and in Chapter Two I indicated that fish passage at a ramp depended on water velocities. Therefore, if water velocities can be kept within the swimming ability of the target fish species with as short a ramp as possible, upstream passage at ramps should improve. To provide passage for inanga, velocities should ideally be below 0.3 m/s for adults and below 0.2 m/s for juveniles (Stancliff *et al.* 1988; Boubée *et al.* 1999). Larger fish are able to swim faster and longer, independently of their life stage (Chapter Two) and many diadromous fish species, such as salmonids and galaxiids, commence their migration as juveniles (Nikora *et al.* 2003; Ojanguren & Brana 2003; Tudorache *et al.* 2008). Ramp designs incorporating target fish size and/or life stage in their planning are hence more likely to improve upstream passage. However, it could be difficult to achieve 'ideal' flow conditions, especially in streams of third order or higher, due to high water flow capacities. In this case, the flow upstream of the ramp could be, for instance, diverted so that only a proportion of it enters the ramp. In general, flow conditions should be as natural as possible and resemble those encountered during migratory periods as these are the times where most fish will try to reach upper stream reaches for growth and reproduction (McDowall 2001).

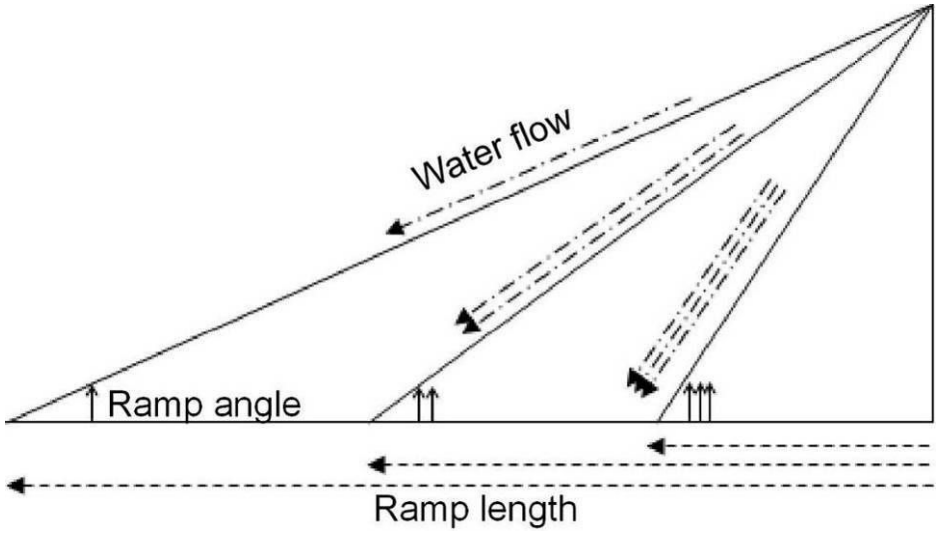


Figure 3. Three possible solutions to the trade-off between water flow, ramp length and ramp angle to facilitate fish passage. The alteration of one variable causes the two other dependent variables to change.

A method to prevent increased velocities and subsequently turbulence at the ramp entrance can be to slow down the flow by rough ramp surface material (Baker & Boubée 2006; Richmond *et al.* 2007). Astroturf proved to be a suitable substrate material in my experiments, however the robustness of the material is a major design consideration and its suitability for long-term *in situ* installations needs to be further investigated. Furthermore, Stripdrain© (a pipe lined with a thin plastic sheet with rows of 24 mm high cones at 30 mm centres and 15 mm spacing at the base) was a suitable substrate to facilitate upstream passage for some New Zealand fish species within culverts as well as on fish ramps (Hegberg *et al.* 2001a; Baker & Boubée 2006; Stevenson *et al.* 2008). Although this material is not particularly rough, it provides diverse flow conditions on the ramp and can facilitate upstream passage for other weak swimming species.

Finally, to maintain the function of fish ramps and culverts, they should be inspected at least once a year, especially prior to or at the start of migration periods (Adams & Whyte 1990). Maintenance generally includes the removal of debris or bedload accumulations on the ramp and/or at the culvert outlet and inlet (Cotterell 1998).

4.4. FUTURE RESEARCH

The main issues of urbanisation on fish distribution have started to be unravelled, but there are still significant gaps in knowledge. Future experiments incorporating, for instance, other fish species or investigating the capability of whitebait to get through culverts, would provide valuable information for the development of passage design, and also significantly increase the amount of information on fish behaviour for a wider variety of target species (MacDonald & Davies 2007). Further studies would also likely improve the understanding and usefulness of pools as resting opportunities on ramps for whitebait species. Studies should ideally include the effectiveness of pools at ramps longer than three metres and investigate potential effects of macro-eddies at pool entrances (Rajaratnam *et al.* 1997; Kim 2001).

The compatibility and practicability of using the surface materials used in my and previous studies also needs to be looked into. For field installations particularly, it needs to be investigated whether Astroturf or Stripdrain© are robust enough to sustain long periods of time under field conditions. Councils frequently prefer materials that are more natural and blend into the surrounding landscape such as rock-ramps. Rock ramps present a viable solution at culverts with fish passage issues (Strickland 2009), but their construction can be complex (i.e., lack of space for the construction and the ramp itself) or cost-intensive for retrofitting of perched culverts (Larnier 2002). Hence, the 'ideal' ramp still needs to be developed. The design developed in my study together with the existing rock-ramp idea could be potentially used to create a "hybrid" ramp that can easily be installed at culverts that need to be retrofitted and are durable.

Finally, urban streams have become important to the public for recreation, aesthetics and as places of cultural and historic value (Suren & Elliott 2004). If they are managed properly, urban streams can also provide places where diverse flora and fauna may be found even within the frame of urban development. However, to be able to restore our urban waterways, and the communities they support, we must identify what can be done to minimise the effects of urbanisation. My study

showed that even minor alterations to conventional urban stream design should result in major improvements in the distribution of freshwater fish species. This can eventually help to enhance or maintain the health of our urban stream ecosystems.

By 2050, 70 % of the world population is expected to live in cities (UN 2007). Hence healthy urban waterways will, in the long run, favour the health of our own kind and we should therefore no longer neglect these unique ecosystems but try and integrate their needs into our modern way of life.

REFERENCES

Adams M. A. & Whyte I. W. (1990) Fish habitat enhancement - a manual for freshwater, estuarine and marine habitats. Prepared for Government of Canada. *Department of Fisheries and Oceans, Canada*, 330 p.

Allan J. D. (2004) Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annual Review of Ecology, Evolution, and Systematics*, **35**, 257-284.

Allibone R., Caskey D. & Miller R. (2003) Population structure, individual movement, and growth rate of shortjaw kokopu (*Galaxias postvectis*) in two North Island, New Zealand streams. *New Zealand Journal of Marine and Freshwater Research*, **37**, 473-483.

Allibone R. M. (1999) Impoundment and introductions: their impacts on native fish of the upper Waiapori River, New Zealand. *Journal of the Royal Society of New Zealand*, **29**, 291-299.

Allibone R. M. & Caskey D. (2000) Timing and habitat of koaro (*Galaxias brevipinnis*) spawning in streams draining Mt Taranaki, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **34**, 593-595.

Baker C. F. (2003) Effect of fall height and notch shape on the passage of inanga (*Galaxias maculatus*) and common bullies (*Gobiomorphus cotidianus*) over an experimental weir. *New Zealand Journal of Marine and Freshwater Research*, **37**, 283-290.

Baker C. F. & Boubée J. A. T. (2006) Upstream passage of inanga (*Galaxias maculatus*) and redfin bullies (*Gobiomorphus huttoni*) over artificial ramps. *Journal of Fish Biology*, **69**, 668-681.

Baker C. F. & Hicks B. J. (2003) Attraction of migratory inanga (*Galaxias maculatus*) and koaro (*Galaxias brevipinnis*) juveniles to adult galaxiid odours. *New Zealand Journal of Marine and Freshwater Research*, **37**, 291-299.

Baker C. F. & Montgomery J. C. (2001) Species-specific attraction of migratory banded kokopu juveniles to adult pheromones. *Journal of Fish Biology*, **58**, 1221-1229.

Baker C. F. & Smith J. (2007) Habitat use by banded kokopu (*Galaxias fasciatus*) and giant kokopu (*G. argenteus*) co-occurring in streams draining the Hakarimata Range, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **41**, 25-33.

Baras E. & Cherry B. (1990) Seasonal activities of female barbel *Barbus barbus* (L.) in the River Ourthe (Southern Belgium) as revealed by radio-tracking. *Aquatic Living Resources*, **3**, 283-294.

Beamish F. W. H. (1970) Oxygen consumption of largemouth bass, *Micropterus salmoides*, in relation to swimming speed and temperature. *Canadian Journal of Zoology*, **48**, 1221-1228.

Beatty S. J., Morgan D. L. & Torre A. (2007) Restoring ecological connectivity in the Margaret River: western Australia's first rock-ramp fishways. *Ecological Management and Restoration*, **8**, 224-228.

Bender M. J., Katopodis C. & Simonovic S. P. (1992) A prototype expert system for fishway design. *Environmental Monitoring and Assessment*, **23**, 115-127.

Bernhardt E. S. & Palmer M. A. (2007) Restoring streams in an urbanizing world. *Freshwater Biology*, **52**, 738-751.

Blakely T. J. & Harding J. S. (2005) Longitudinal patterns in benthic communities in an urban stream under restoration. *New Zealand Journal of Marine and Freshwater Research*, **39**, 17-28.

Blakely T. J., Harding J. S., McIntosh A. R. & Winterbourn M. J. (2006) Barriers to the recovery of aquatic insect communities in urban streams. *Freshwater Biology*, **51**, 1634-1645.

Bonnett M. L. & Sykes J. R. E. (2002) Habitat preferences of giant kokopu, *Galaxias argenteus*. *New Zealand Journal of Marine and Freshwater Research*, **36**, 13-24.

Boubée J. A., Jowett I. G., Nichols S. & Williams E. (1999) Fish passage at culverts: a review, with possible solutions for New Zealand indigenous species. A report for the Department of Conservation, New Zealand. NIWA client report, Wellington, 29p.

Boubée J. A., Mitchell C. P., Chisnall B. L., West D. W., Bowman E. J. & Haro A. (2001) Factors regulating the downstream migration of mature eels (*Anguilla* spp.) at Aniwhenua Dam, Bay of Plenty, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **35**, 121-134.

Boubée J. A. T. & Williams E. K. (2006) Downstream passage of silver eels at a small hydroelectric facility. *Fisheries Management and Ecology*, **13**, 165-176.

Bunt C. M. (2001) Fishway entrance modifications enhance fish attraction. *Fisheries Management and Ecology*, **8**, 95-105.

Burnham K. P. & Anderson D. R. (2002) *Model selection and multimodel inference: a practical information-theoretic approach.* Springer, New York, 488p.

Castro-Santos T., Haro A. & Walk S. (1996) A passive integrated transponder (PIT) tag system for monitoring fishways. *Fisheries Research*, **28**, 253-261.

Cea L., Pena L., Puertas J., Vázquez-Cendón M. E. & Peña E. (2007) Application of several depth-averaged turbulence models to simulate flow in vertical slot fishways. *Journal of Hydraulic Engineering*, **133**, 160-172.

Charteris S. C., Allibone R. M. & Death R. G. (2003) Spawning site selection, egg development, and larval drift of *Galaxias postvectis* and *G. fasciatus* in a New Zealand stream. *New Zealand Journal of Marine and Freshwater Research*, **37**, 493-505.

Clay C. H. (1995) *Design of fishways and other fish facilities*, Lewis Publishers, Ottawa, Canada, 248p.

Cotterell E. (1998) Fish passage in streams - fisheries guidelines for design of stream crossings. *Fish Habitat Guidelines FHG 001,* Queensland Department of Primary Industries DPI, 40p.

Coutant C. C. & Whitney R. R. (2000) Fish Behaviour in relation to passage through hydropower turbines: a review. *Transactions of the American Fisheries Society*, **129**, 351-380.

Crawley M. J. (2007) *The R book*. Wiley, Chichester, UK, 950p.

Crowe A. (2002) Surface water quality in the Nelson region 2000/2001. Cawthron Report, **685**, 84p.

Dean T. L. & Richardson J. (1999) Responses of seven species of native freshwater fish and a shrimp to low levels of dissolved oxygen. *New Zealand Journal of Marine and Freshwater Research*, **33**, 99-106.

Dodd C. K., Barichivich W. J. & Smith L. L. (2004) Effectiveness of a barrier wall and culverts in reducing wildlife mortality on a heavily travelled highway in Florida. *Biological Conservation*, **118**, 619-631.

Doehring K. A. M. & McIntosh A. R. (2008) The distribution of native freshwater fish in the Nelson City Council area. A report for the Nelson City Council. Freshwater Ecology Research Group, School of Biological Sciences, University of Canterbury, Christchurch, New Zealand, 36p.

Dominy C. L. (1973) Effect of entrance-pool weir elevation and fish density on passage of alewives (*Alosa pseudoharengus*) in a pool and weir fishway. *Transactions of the American Fisheries Society*, **2**, 398-404.

Eikaas H. S., Kliskey A. D. & McIntosh A. R. (2006) Analysis of patterns in diadromous fish distributions using GIS. *Transactions in GIS*, **10**, 469-483.

Eikaas H. S. & McIntosh A. R. (2006) Habitat loss through disruption of constrained dispersal networks. *Ecological Applications*, **16**, 987-998.

Eikaas H. S., McIntosh A. R. & Kliskey A. D. (2005) Catchment- and site-scale influences of forest cover and longitudinal forest position on the distribution of a diadromous fish. *Freshwater Biology*, **50**, 527-538.

ESRI (2006) ESRI Arc GIS 9.2, ESRI inc., Redlands, CA, USA.

European Parliament and Council (2000) Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000. The European Parliament and the Council of the European Union, 82p.

Faragher R. A. & Harris J. H. (1994) The historical and current status of freshwater fish in New South Wales. *Australian Zoologist*, **29**, 166-176.

Fitch G. M. (1996) Avoidance of nonanadromous fish passage impedance caused by highway culverts. *Transportation Research Record*, **1559**, 34-41.

Folke C., Jansson A., Larsson J. & Costanza R. (1997) Ecosystem appropriation by cities. *Ambio*, **26**, 167-172.

Gibson C. A., Meyer J. L., Poff N. L., Hay L. E. & Georgakakos A. (2005a) Flow regime alterations under changing climate in two river basins: implications for freshwater ecosystems. *River Research and Applications*, **21**, 849-864.

Gibson R. J., Haedrich R. L. & Wernerheirn C. M. (2005b) Loss of fish habitat as a consequence of inappropriately constructed stream crossings. *Fisheries*, **30**, 10-17.

Glova G. J., Jellyman D. J. & Bonnett M. L. (1998) Factors associated with the distribution and habitat of eels (*Anguilla* spp.) in three New Zealand lowland streams. *New Zealand Journal of Marine and Freshwater Research*, **32**, 255-269.

Griffiths D. (2006) Pattern and process in the ecological biogeography of European freshwater fish. *Journal of Animal Ecology*, **75**, 734-751.

Grimm N. B., Foster D., Groffman P., Grove J. M., Hopkinson C. S., Nadelhoffer K. J., Pataki D. E. & Peters D. P. C. (2008) The changing landscape: Ecosystem responses to urbanization and pollution across climatic and societal gradients. *Frontiers in Ecology and the Environment*, **6**, 264-272.

Grimm N. B., Grove J. M., Pickett S. T. A. & Redman C. L. (2000) Integrated approaches to long-term studies of urban ecological systems. *BioScience*, **50**, 571-584.

Haggerty J. H. (2007) "I'm not a greenie but...": environmentality, eco-populism and governance in New Zealand: experiences from the Southland whitebait fishery. *Journal of Rural Studies*, **23**, 222-237.

Hammer C. (1995) Fatigue and exercise tests with fish. *Comparative Biochemistry and Physiology*, **112**, 1-20.

Hanchet S. M. (1990) Effect of land use on the distribution and abundance of native fish in tributaries of the Waikato River in the Hakarimata Range, North Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **24**, 159-171.

Harding J. M., Burky A. J. & Way C. M. (1998) Habitat preferences of the rainbow darter, *Etheostoma caeruleum*, with regard to microhabitat velocity shelters. *Copeia*, **4**, 988-997.

Haro A., Castro-Santos T., Noreika J. & Odeh M. (2004) Swimming performance of upstream migrant fishes in open-channel flow: a new approach to predicting passage through velocity barriers. *Canadian Journal of Fisheries and Aquatic Sciences*, **61**, 1590-1601.

Harvey B. C. (1991) Interactions among stream fishes: predator-induced habitat shifts and larval survival. *Oecologia*, **87**, 29-36.

Hayes J. W., Leathwick J. R. & Hanchet S. M. (1989) Fish distribution patterns and their association with environmental factors in the Mokau River catchment, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **23**, 171-180.

Hegberg C. H., Cohen S. J., Schlindwein P. A. & Jacobs S. (2001a) Natural fish passage structures in urban streams (Part 1: Hydrologic and resource issues). Road Ecology Center eScholarship Repository John Muir Institute of the Environment, University of California, Davis, 108p.

Hegberg C. H., Jacobs S., Schlindwein P. A. & Cohen S. J. (2001b) Fish Passage in urban streams (Part 1: Regulatory and resource issues). Conference Proceedings of the 2001 Wetlands Engineering and River Restoration, 194p.

Hicks B. J. & McCaughan H. M. C. (1997) Land use, associated eel production, and abundance of fish and crayfish in streams in Waikato, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **31**, 635-650.

Hitchmough R., Bull L. & Cromarty P. (2007) New Zealand threat classification system lists 2005. Department of Conservation, Wellington, New Zealand, 134p.

Hogg I. D. & Norris R. H. (1991) Effects of runoff from land clearing and urban development on the distribution and abundance of macroinvertebrates in pool areas of a river. *Australian Journal of Marine & Freshwater Research*, **42**, 507-518.

House M. R., Pyles M. R. & White D. (2005) Velocity distributions in streambed simulation culverts used for fish passage. *Journal of the American Water Resources Association*, **41**, 209-217.

Hudson R. C. L. (1973) On the function of the white muscles in teleosts at intermediate swimming speeds. *Journal of Experimental Biology*, **58**, 509-522.

James A. & Joy M. (2008) A preliminary assessment of potential barriers to fish migration in the Manawatu River catchment, North Island, New Zealand. A report for the Horizons (Manawatu-Wanganui) Regional Council, Massey University, 137p.

Johnson P. A. & Brown E. R. (2000) Stream assessment for multicell culvert use. *Journal of Hydraulic Engineering*, **126**, 381-386.

Jowett I. G. (2002) In-stream habitat suitability criteria for feeding inanga (*Galaxias maculatus*). *New Zealand Journal of Marine and Freshwater Research*, **36**, 399-407.

Jowett I. G. & Richardson J. (1994) Comparison of habitat use by fish in normal and flooded river conditions. *New Zealand Journal of Marine and Freshwater Research*, **28**, 409-416.

Jowett I. G. & Richardson J. (1996) Distribution and abundance of freshwater fish in New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research*, **30**, 239-255.

Jowett I. G., Richardson J. & McDowall R. M. (1996) Relative effects of in-stream habitat and land use on fish distribution and abundance in tributaries of the Grey River, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **30**, 463-475.

Joy M. K. & Death R. G. (2000) Development and application of a predictive model of riverine fish community assemblages in the Taranaki region of the North Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **34**, 241-252.

Joy M. K. & Death R. G. (2004) Predictive modelling and spatial mapping of freshwater fish and decapod assemblages using GIS and neural networks. *Freshwater Biology*, **49**, 1036-1052.

Joy M. K., Henderson I. M. & Death R. G. (2000) Diadromy and longitudinal patterns of upstream penetration of freshwater fish in Taranaki, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **34**, 531-543.

Kemp P. S. & Williams J. G. (2008) Response of migrating Chinook salmon (*Oncorhynchus tshawytscha*) smolts to in-stream structure associated with culverts. *River Research and Applications*, **24**, 571-579.

Kennard M. J., Arthington A. H., Pusey B. J. & Harch B. D. (2005) Are alien fish a reliable indicator of river health? *Freshwater Biology*, **50**, 174-193.

Kim J. H. (2001) Hydraulic characteristics by weir type in a pool-weir fishway. *Ecological Engineering*, **16**, 425-433.

Knaepkens G., Baekelandt K. & Eens M. (2006) Fish pass effectiveness for bullhead (*Cottus gobio*), perch (*Perca fluviatilis*) and roach (*Rutilus rutilus*) in a regulated lowland river. *Ecology of Freshwater Fish*, **15**, 20-29.

Knaepkens G., Bruyndoncx L. & Eens M. (2004) Assessment of residency and movement of the endangered bullhead (*Cottus gobio*) in two Flemish rivers. *Ecology of Freshwater Fish*, **13**, 317-322.

Lake P. S., Bond N. & Reich P. (2007) Linking ecological theory with stream restoration. *Freshwater Biology*, **52**, 597-615.

Larnier M. (2002) Fish passage through culverts, rock weirs and estuarine obstructions. *Bull. Fr. Pêche Piscic.*, **364**, 119-134.

Leathwick J. R., Elith J., Chadderton W. L., Rowe D. & Hastie T. (2008) Dispersal, disturbance and the contrasting biogeographies of New Zealand's diadromous and non-diadromous fish species. *Journal of Biogeography*, **35**, 1481-1497.

Liao J. C., Beal D. N., Lauder G. V. & Triantafyllou M. S. (2003a) Fish exploiting vortices decrease muscle activity. *Science*, **302**, 1566-1569.

Liao J. C., Beal D. N., Lauder G. V. & Triantafyllou M. S. (2003b) The Kármán gait: novel body kinematics of rainbow trout swimming in a vortex street. *Journal of Experimental Biology*, **206**, 1059-1073.

Lozon J. D. & MacIsaac H. J. (1997) Biological invasions: are they dependent on disturbance? *Environmental Reviews*, 131-144.

Lucas M. C., Mercer T., Armstrong J. D., McGinty S. & Rycroft P. (1999) Use of a flat-bed passive integrated transponder antenna array to study the migration and behaviour of lowland river fishes at a fish pass. *Fisheries Research*, **44**, 183-191.

MacDonald J. I. & Davies P. E. (2007) Improving the upstream passage of two galaxiid fish species through a pipe culvert. *Fisheries Management and Ecology*, **14**, 221-230.

Magaud H., Migeon B., Morfin P., Garric J. & Vindimian E. (1997) Modelling fish mortality due to urban storm run-off: Interacting effects of hypoxia and un-ionized ammonia. *Water Research*, **31**, 211-218.

Marchetti M. P., Light T. S., Feliciano J., Armstrong T. W., Hogan Z. & Moyle P. B. (2001) Physical homogenization and biotic homogenization in aquatic systems. In: *Biotic homogenization* (ed. Lockwood J.L., McKinney, M.L.), pp. 259–278. Kluwer/Academic Press, New York.

Marchetti M. P., Lockwood J. L. & Light T. (2006) Effects of urbanization on California's fish diversity: differentiation, homogenization and the influence of spatial scale. *Biological Conservation*, **127**, 310-318.

Mardones A., Vega R. & Encina F. (2008) Cultivation of whitebait (*Galaxias maculatus*) in Chile. *Aquaculture Research*, **39**, 731-737.

McCune & Grace (2002) *Analysis of ecological communities.* MjM Software, Gleneden Beach, OR, U.S.A, 284p.

McDowall R. M. (1993) Implications of diadromy for the structuring and modelling of riverine fish communities in New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **27**, 453-462.

McDowall R. M. (1995) Seasonal pulses in migrations of New Zealand diadromous fish and the potential impacts of river mouth closure. *New Zealand Journal of Marine and Freshwater Research*, **29**, 517-526.

McDowall R. M. (1997) Indigenous vegetation type and the distribution of shortjawed kokopu, *Galaxias postvectis* (Teleostei: Galaxiidae), New Zealand. *New Zealand Journal of Zoology*, **24**, 243-255.

McDowall R. M. (1998) Fighting the flow: downstream-upstream linkages in the ecology of diadromous fish faunas in West Coast New Zealand rivers. *Freshwater Biology*, **40**, 111-122.

McDowall R. M. (2000) *The Reed field guide to New Zealand freshwater fishes*, Reed Publishing (NZ) Ltd., Auckland, 224p.

McDowall R. M. (2001) *Freshwater Fishes of New Zealand*, Nature Series, Reed Publishing (NZ) Ltd., Auckland, 230p.

McDowall R. M. (2003) The key to climbing in the koaro. *NIWA Water & Atmosphere*, **11**, 16-17.

McDowall R. M. (2006) Crying wolf, crying foul, or crying shame: alien salmonids and a biodiversity crisis in the southern cool-temperate galaxioid fishes? *Reviews in Fish Biology and Fisheries*, **16**, 233-422.

McDowall R. M. (2007) On amphidromy, a distinct form of diadromy in aquatic organisms. *Fish and Fisheries*, **8**, 1-13.

McDowall R. M. & Eldon G. A. (1980) The ecology of whitebait migrations (Galaxiidae: *Galaxias* spp.). New Zealand Ministry of Agriculture and Fisheries, Christchurch, 89p.

McDowall R. M. & Taylor M. J. (2000) Environmental indicators of habitat quality in a migratory freshwater fish fauna. *Environmental Management*, **25**, 357-374.

McIntosh A. R. & McDowall R. M. (2004) Fish communities in rivers and streams. In: *Freshwaters of New Zealand* (eds. Harding J., Mosley P., Pearson C. & Sorell B.), pp. 17.11 -17.20. New Zealand Hydrological and Limnological Societies, Christchurch, New Zealand.

McIntosh A. R., Townsend C. R. & Crowl T. A. (1992) Competition for space between introduced brown trout (*Salmo trutta*) and common river galaxias (*Galaxias vulgaris*) in a New Zealand stream. *Journal of Fish Biology*, **41**, 63-81.

Miltner R. J., White D. & Yoder C. (2004) The biotic integrity of streams in urban and suburbanizing landscapes. *Landscape and Urban Planning*, **69**, 87-100.

Ministry for the Environment, The New Zealand Land Cover Database, *http://www.mfe.govt.nz/issues/land/land-cover-dbase/*, 11.06.09.

Minns C. K. (1990) Patterns of distribution and association of freshwater fish in New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **24**, 31-44.

Mitchell C. P. (1989) Swimming performances of some native freshwater fishes. *New Zealand Journal of Marine and Freshwater Research*, **23**, 181-187.

Myers G. S. (1949) Usage of anadromous, catadromous and allied terms for migratory fishes. *Copeia*, **1949**, 89-97.

Nagashima K., Sands R., Whyte A. G. D., Bilek E. M. & Nakagoshi N. (2003) Plantation expansion possibility and its influence on land-use pattern in the Nelson region, New Zealand. *Forest Ecology and Management*, **184**, 263-275.

National Institute of Water & Atmospheric Research (NIWA), New Zealand Freshwater Fish Database, *http://fwdb.niwa.cri.nz*, 25.05.2008.

Naughton G. P., Caudill C. C., Peery C. A., Clabough T. S., Jepson M. A., Bjornn T. C. & Stuehrenberg L. C. (2007) Experimental evaluation of fishway modifications on the passage behaviour of adult Chinook salmon and steelhead at Lower Granite Dam, Snake River, USA. *River Research and Applications*, **23**, 99-111.

Nelson City Council, About Nelson - By the numbers 2001, *http://www.nelsoncitycouncil.co.nz/aboutnelson/nelsoninfocus/bynumbers.htm*, 12.06.2009.

Nelson E. J. & Booth D. B. (2002) Sediment sources in an urbanizing, mixed land-use watershed. *Journal of Hydrology*, **264**, 51-68.

Nikora V. I., Aberle J., Biggs B. J. F., Jowett I. G. & Sykes J. R. E. (2003) Effects of fish size, time-to-fatigue and turbulence on swimming performance: a case study of *Galaxias maculatus*. *Journal of Fish Biology*, **63**, 1365-1382.

O'Hanley J. R. & Tomberlin D. (2005) Optimizing the removal of small fish passage barriers. *Environmental Modeling and Assessment*, **10**, 85-98.

Ojanguren A. F. & Brana F. (2003) Effects of size and morphology on swimming performance in juvenile brown trout (*Salmo trutta* L.). *Ecology of Freshwater Fish*, **12**, 241-246.

Parker M. A. (2000) Fish passage - culvert inspection procedures. *Watershed Restoration Technical Circular*, **11**, 1-47.

Paul M. J. & Meyer J. L. (2001) Streams in the urban landscape. *Annual Review of Ecology and Systematics*, **32**, 333-365.

Pavlov D. S., Mikheev V. N., Lupandin A. I. & Skorobogatov M. A. (2008) Ecological and behavioural influences on juvenile fish migrations in regulated rivers: a review of experimental and field studies. *Hydrobiologia*, **609**, 125-138.

Peake S. (2004) An evaluation of the use of critical swimming speed for determination of culvert water velocity criteria for smallmouth bass. *Transactions of the American Fisheries Society*, **133**, 1472-1479.

Peake S., McKinley R. S. & Scruton D. A. (1997a) Swimming performance of various freshwater Newfoundland salmonids relative to habitat selection and fishway design. *Journal of Fish Biology*, **51**, 710-723.

Peake S. J., Beamish F. W. H., McKinley R. S., Scruton D. A. & Katapodis C. (1997b) Relating swimming performance of lake sturgeon, *Acipenser fulvescens*, to fishway design. *Canadian Journal of Fisheries and Aquatic Sciences*, **54**, 1361-1366.

Plew D. R., Nikora V. I., Larned S. T., Sykes J. R. E. & Cooper G. G. (2007) Fish swimming speed variability at constant flow: *Galaxias maculatus*. *New Zealand Journal of Marine and Freshwater Research*, **41**, 185-195.

Pringle C. M. (2001) Hydrologic connectivity and the management of biological reserves: a global perspective. *Ecological Applications*, **11**, 981-998.

R Foundation, The R Foundation for statistical computing 2007, *http://www.r-project.org*, 2.6. R, 19.02.2009.

Rajaratnam N., Katopodis C., Wu S. & Sabur M. A. (1997) Hydraulics of resting pools for Denil fishways. *Journal of Hydraulic Engineering*, **123**, 632-638.

Reynolds L. F. (1983) Migration patterns of five fish species in the Murray-Darling River system. *Australian Journal of Marine & Freshwater Research*, **34**, 857-871.

Richardson J. & Jowett I. G. (2002) Effects of sediment on fish communities in East Cape streams, North Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **36**, 431-442.

Richmond M. C., Deng Z., Guensch G. R., Tritico H. & Pearson W. H. (2007) Mean flow and turbulence characteristics of a full-scale spiral corrugated culvert with implications for fish passage. *Ecological Engineering*, **30**, 333-340.

Rowe D., Hicks M. & Richardson J. (2000) Reduced abundance of banded kokopu (*Galaxias fasciatus*) and other native fish in turbid rivers of the North Island of New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **34**, 547-556.

Rowe D. K., Chisnall B. L., Dean T. L. & Richardson J. (1999) Effects of land use on native fish communities in east coast streams of the North Island of New Zealand. *New Zealand Journal of Marine and Freshwater Research*, **33**, 141-151.

Rowe D. K. & Dean T. L. (1998) Effects of turbidity on the feeding ability of the juvenile migrant stage of six New Zealand freshwater fish species. *New Zealand Journal of Marine and Freshwater Research*, **32**, 21-29.

Rowe D. K., Saxton B. A. & Stancliff A. G. (1992) Species composition of whitebait (Galaxiidae) fisheries in twelve Bay of Plenty rivers, New Zealand: evidence for river mouth selection by juvenile *Galaxias brevipinnis* (Günther). *New Zealand Journal of Marine and Freshwater Research*, **26**, 219-228.

Rowe D. K., Smith J. & Williams E. (2002) Effects of turbidity on the feeding ability of adult, riverine smelt (*Retropinna retropinna*) and inanga (*Galaxias maculatus*). *New Zealand Journal of Marine and Freshwater Research*, **36**, 143-150.

Schilt C. R. (2007) Developing fish passage and protection at hydropower dams. *Applied Animal Behaviour Science* **104**, 295–325.

Scott M. C. (2006) Winners and losers among stream fishes in relation to land use legacies and urban development in the southeastern US. *Biological Conservation*, **127**, 301-309.

Snelder T., Dey K. & Leathwick J. R. (2005) Definition of the multivariate environmental river classification - "Freshwater environments of New Zealand". NIWA client report, 86p.

Stancliff A. G., Boubée J. A., Palmer D. & Mitchell C. P. (1988) The upstream migration of whitebait species in the Lower Waikato River. New Zealand Freshwater Fisheries Report. Ministry of Agriculture and Fisheries, **96**, 44p.

Statistics New Zealand, Statistics New Zealand (2007), Quarterly regional reviews - December 2007, *www.stats.govt.nz/statistics-by-area/regional-statistics/quarterly-reg-review/December+2007+quarter.htm,* 29.05.2008.

Stevenson C., Kopeinig T., Feurich R. & Boubée J. (2008) Culvert barrel design to facilitate the upstream passage of small fish. A report for the Auckland Regional Council. NIWA client report, 49p.

Strickland R., Fish passage in New Zealand rivers, *http://www.cawthron.org.nz/coastal-freshwater-resources/downloads/fish-passage.pdf,* The Cawthron Institute, 14.06.2009.

Stuart I. G. & Mallen-Cooper M. (1999) An assessment of the effectiveness of a vertical-slot fishway for non-salmonid fish at a tidal barrier on a large tropical/subtropical river. *Regulated Rivers: Research & Management*, **15**, 575-590.

Suren A. & Elliott S. (2004) Impacts of urbanisation on streams. In: *Freshwaters of New Zealand* (eds. Harding J., Mosley P., Pearson C. & Sorell B.), pp. 35.1 - 35.18. New Zealand Hydrological and Limnological Societies, Christchurch, New Zealand.

Townsend C. R. & Crowl T. A. (1991) Fragmented population structure in a native New Zealand fish: an effect of introduced brown trout. *Oikos*, **61**, 347-254.

Tudorache C., Viaene P., Blust R., Vereecken H. & De Boeck G. (2008) A comparison of swimming capacity and energy use in seven European freshwater fish species. *Ecology of Freshwater Fish*, **17**, 284-291.

UN P. D., World Urbanization Prospects: the 2007 Revision, *http://www.un.org/esa/population/publications/wup2007/2007_urban_agglomerations_chart.pdf,* 23.02.2009.

UN S. D., Agenda 21. United Nations Conference on Environment & Development Rio de Janerio, Brazil, 3 to 14 June 1992, *http://www.un.org/esa/sustdev/documents/agenda21/english/Agenda21.pdf,* 02.06.09.

Walton B. M., Salling M., Wyles J. & Wolin J. (2007) Biological integrity in urban streams: towards resolving multiple dimensions of urbanisation. *Landscape and Urban Planning*, **79**, 110-123.

Wang L., Lyons J., Kanehl P. & Bannerman R. (2001) Impacts of urbanization on stream habitat and fish across multiple spatial scales. *Environmental Management*, **28**, 255-266.

Wargo R. S. & Weisman R. N. (2006) A comparison of single-cell and multicell culverts for stream crossings. *Journal of the American Water Resources Association*, **42**, 989-995.

Warren M. L. (1998) Road crossings as barriers to small-stream fish movement. *Transactions of the American Fisheries Society*, **127**, 637-644.

Webb P. W. (1998) Entrainment by river chub *Nocomis micropogon* and smallmouth bass *Micropterus dolomieu* on cylinders. *Journal of Experimental Biology*, **201**, 2403-2412.

Webb P. W. & Gerstner C. L. (2000) Fish swimming behaviour: predictions from physical principles. In: *Biomechanics in Animal Behaviour*, pp. 59-77. BIOS Scientific, Oxford.

Weihs D. (1974) Energetic advantages of burst swimming of fish. *Journal of Theoretical Biology*, **48**, 215-229.

Young R. G. & Clark K. (2006) Native fish distribution in the Nelson City Council Area. A report for the Nelson City Council. Cawthron Report, **1241**, 11.